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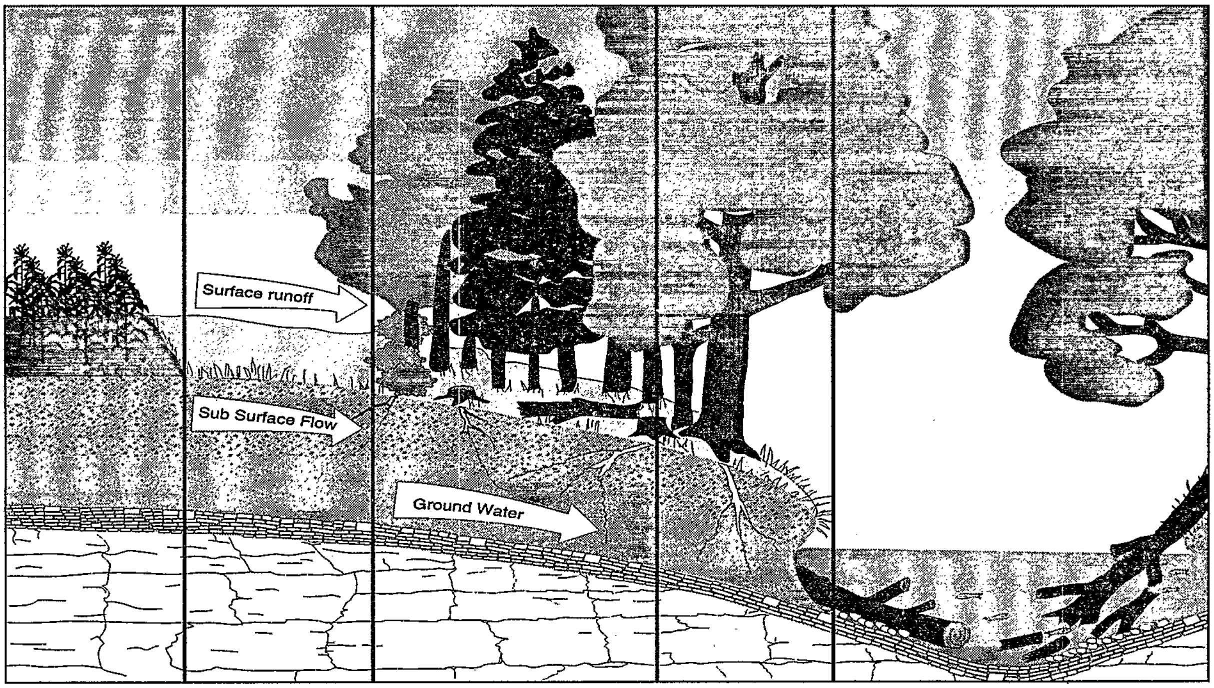
Technology Transfer Report

Water Quality Functions of Riparian Forest Buffer Systems in the Chesapeake Bay Watershed



Prepared by the
Nutrient Subcommittee
of the
Chesapeake Bay Program

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A Report of the
Nutrient Subcommittee
of the Chesapeake Bay Program

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Preface

This document is a research synthesis requested by the Forestry Work Group of the Nutrient Subcommittee of the Chesapeake Bay Program. In developing the outline for the report, the authors agreed to specifically focus on the existing Riparian Forest Buffer System (RFBS) specification developed by USDA and being used as a starting point for federal, state, and local RFBS specifications. Although the report contains a general review of riparian forest and grass vegetated filter strip literature, the goal was to use this literature to help determine the applicability of the forest buffer system recommended by USDA.

The strategy for development of the document was to bring together researchers in this field to: 1) discuss the current state of knowledge of RFBS; 2) determine how that knowledge related to the Chesapeake Bay Watershed; and 3) reach consensus about the functions of RFBS in the Bay watershed based on that current state of knowledge. The consensus statements are very important but they do not ensure specific functions will result from RFBS in a given field setting. Rather, they are Best Professional Judgements of the entire authors group and represent general agreement among the authors about the current validity of the statements. In addition to the authors, a large number of reviewers were asked to examine the report and form their own judgements about the general conclusions. These reviewers, acknowledged below, generally agree with the consensus statements contained in the report.

As readers of this report will see, numerous scientific questions remain about the role of RFBS in all of the physiographic and land use settings of the Bay watershed. Yet, incomplete scientific knowledge can not be used to avoid making informed management judgements, especially when society has determined that a globally important natural resource such as Chesapeake Bay must be restored to ecological health. The scientists involved with the preparation of this report have attempted to make the best judgements possible to help guide the application of RFBS to improve water quality in the Chesapeake Bay Watershed and ultimately in the Bay itself.

Acknowledgements

The authors are especially grateful to Mr. Spencer Waller, who provided logistical support for our meetings and provided detailed minutes of the discussions. In particular, Spencer transcribed about 25 hours of discussions which included our process of reaching consensus on the applicability of RFBS in different physiographic settings. Without these direct transcripts, we would not have been able to adequately recapture these statements for the report.

The authors also wish to thank the reviewers of the report. Most of them spent a large amount of time on their review and their comments were very helpful in completing the final draft. The list of reviewers includes many scientists and natural resource managers active in the Chesapeake Bay Watershed. The reviewers were Andrew Dolloff (USDA-FS), Richard Everett (U.S. Fish and Wildlife Service), Verna Harrison (Maryland Dept. of Natural Resources), Thomas Jordan (Smithsonian Environmental Research Center), Larry Lubbers (Maryland Dept. of Natural Resources), Kent Mountford (U.S. Environmental Protection Agency), Adel Shirmohammadi (University of Maryland), George Simmons (Virginia Polytechnic Institute & State University), Thomas Simpson (Maryland Dept. of Agriculture), Bernard Sweeny (Stroud Water Research Center), and Donald Weller (Smithsonian Environmental Research Center).

A number of people participated in at least one of our two meetings but are not co-authors of the report. Among these people who provided useful input at the meetings were Ed Corbett (USDA-Forest Service), Rupert Friday (Chesapeake Bay Foundation), Bob Merrill (Pennsylvania Bureau of Forestry), Kent Mountford (U.S. Environmental Protection Agency), Ann Swanson (Chesapeake Bay Commission), Bob Tjaden (Univ. of Maryland), and Dave Welsch (USDA-FS).

As with any undertaking of this sort, much of the work was done by people who get little credit. Ms. Olive Sides, Office Automation Assistant with USDA-ARS, Tifton, GA helped with much of the correspondence and arrangements for the two meetings held to develop the report. Ms. Dalma Dickens, Secretary with USDA-ARS, Tifton, typed numerous versions of the report. Mr. H. L. Batten, USDA-ARS, Tifton, Ms. Wendy R. Pierce, USDA-NRCS, Ft. Worth, Texas, and Mr. Anthony J. Kimmit, USDA-NRCS, Ft. Worth, Texas prepared many of the figures.

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Executive Summary

Riparian (streamside) forests are known to reduce delivery of nonpoint source pollution to streams and lakes in many types of watersheds. In addition, riparian forests are known to be important in controlling the physical and chemical environment of streams and in providing detritus and woody debris for streams and near-shore areas of water bodies. Riparian forests were the original native vegetation in most streamside areas of the Chesapeake Bay Watershed. This report assesses the state of scientific knowledge concerning the Water Quality Functions of riparian ecosystems. This assessment and specific knowledge of riparian ecosystem function in physiographic regions of the Chesapeake Bay Watershed were used to make consensus Best Professional Judgements as to the potential water quality functions of Riparian Forest Buffer Systems (RFBS) in the Bay Watershed.

Research conducted in naturally occurring riparian forests and experimental and on-farm grass filter strips has been used by the U.S. Department of Agriculture to develop a general "Riparian Forest Buffer System specification" for controlling nonpoint source pollution from agriculture and improving general water quality. The specification calls for a three zone buffer system, with each zone having specific purposes but also having interactions with the adjacent zones to provide the overall RFBS function. Zone 1 of the RFBS is an area of permanent forest vegetation immediately adjacent to the stream channel and encompassing at least the entire stream channel system. Zone 2 is an area of managed forest, upslope from Zone 1. Zone 2 is managed for control of pollutants in subsurface flow and surface runoff through biological and chemical transformations, storage in woody vegetation, infiltration, and sediment deposition. Zone 3 is a grass or other herbaceous filter strip upslope from Zone 2. Zone 3 is managed to provide spreading of concentrated flow into sheet flow and to remove sediment and sediment associated pollutants.

The most general function of Riparian Forest Buffer Systems is to provide control of the stream environment. These functions include modifying stream temperature and controlling light quantity and quality; enhancing habitat diversity; modifying channel morphology; and enhancing food webs and species richness. All of these factors are important to the ecological health of a stream and are best provided by a RFBS which includes a Zone 1 that approximates the original native vegetation. These functions occur along smaller streams regardless of physiographic region. These functions are most important on smaller streams, although they are important for bank and near-shore habitat on larger streams and the shoreline of the Bay. RFBS contribute to bank stability and thus minimize sediment loading due to instream bank erosion. Depending on bank stability and soil conditions in Zone 1, management of Zone 2 for long-term rotations may be necessary for sustainability of stream environment functions of Zone 1.

The next most general function of RFBS is control of sediment and sediment-borne pollutants carried in surface runoff. Properly managed RFBS should provide a high level of control of sediment and sediment borne chemicals regardless of physiographic region. Natural riparian forest studies indicate that forests are particularly effective in filtering fine sediments and promoting co-deposition of sediment as water infiltrates. The slope of the RFBS is the main factor limiting the effectiveness of the sediment removal function. In all physiographic settings it is important to convert concentrated flow to sheet flow in order to optimize RFBS function. Conversion to sheet flow and deposition of coarse sediment which could damage young vegetation are the primary functions of Zone 3—the grass vegetated filter strip.

The next most general function of RFBS is to control nitrate in shallow groundwater moving toward streams. When groundwater moves in short, shallow flow paths, such as in the Inner Coastal Plain (primarily the western shore), 90% of the nitrate input may be removed. In contrast, nitrate removal may be minimal in areas where water moves to regional groundwater such as in Piedmont and Valley and Ridge areas with marble or limestone bedrock, respectively. In these and some Outer Coastal Plain regions, high nitrate groundwater may emerge in stream channels and bypass most of the RFBS. In the areas where this occurs or where high nitrate water moves out in seepage faces, deeply rooted trees in Zone 1 or in seepage areas will be essential. The degree to which nitrate (or other groundwater pollutants) will be removed in the RFBS depends on the proportion of groundwater moving in or near the biologically active root zone and on the residence time of the groundwater in these biologically active areas.

The least general function of RFBS appears to be control of dissolved phosphorus in surface runoff or shallow groundwater. Control of sediment-borne P is generally effective. In certain situations, dissolved P can contribute a substantial amount of total P load. Most of the soluble P is bioavailable, so the potential impact of a unit of dissolved P on aquatic ecosystems is greater. It appears that natural riparian forests have very low net dissolved P retention. In managing for increased P retention, effective fine sediment control should be coupled with use of vegetation which can increase P uptake into plant tissue.

Research on functions of natural, restored, and enhanced RFBS is needed in all portions of the Chesapeake Bay Watershed. Research should be directed into four general areas: 1) assessment of existing riparian forests relative to the RFBS standard; 2) assessment of potential RFBS restoration for NPS pollution control; 3) assessment of NPS pollution control in pilot restoration and enhancement projects; 4) determine the effects of management factors on both pollution control and control of the stream environment. The research, because of the need to do relatively large scale projects which last for substantial periods of time, should be coordinated with demonstration restoration/ enhancement projects. Some of the major research questions should address the uncertainty associated with the functions discussed above. Research should be directed toward testing the hypotheses concerning which functions of RFBS occur in specific physiographic settings and the specific management conditions under which these functions are likely to be enhanced. In particular, research on the time to recovery of RFBS functions and the processes which control the various functions should be integrated into demonstration projects.

I Water Quality Functions of Riparian Forest Buffer Systems

A. INTRODUCTION

Riparian Forest Buffer Systems (RFBS) are streamside ecosystems, managed for the enhancement of water quality through control of nonpoint source pollution (NPS) and protection of the stream environment. The use of riparian management zones is relatively well established as a Best Management Practice (BMP) for water quality improvement in forestry practices (Comerford et al., 1992), but has been much less widely applied as a BMP in agricultural areas or in urban or suburban settings. RFBS are especially important on small streams where intense interaction between terrestrial and aquatic ecosystems occurs. First and second order streams comprise nearly three-quarters of the total stream length in the United States (Leopold et al., 1964). Fluvial activities influence the composition of riparian plant communities along these small streams (Gregory et al., 1991). Likewise, terrestrial disturbances can have an immediate impact on aquatic populations (Sweeney, 1993; Webster et al., 1992). Small streams can be completely covered by the canopies of streamside vegetation (Sweeney, 1992). Riparian vegetation has well-known beneficial effects on the bank stability, biological diversity, and water temperatures of streams (Karr and Schlosser, 1978). Riparian forests of mature trees (30 to 75 yrs. old) are known to effectively reduce nonpoint pollution from agricultural fields (Lowrance et al., 1985b).

Compared to other NPS pollution control measures, RFBS can lead to longer-term changes in the structure and function of agricultural landscapes. To produce long-term improvements in water quality, RFBS must be designed with an understanding of: 1) the processes which remove or sequester pollutants entering the riparian buffer system; 2) the effects of riparian management practices on pollutant retention; 3) the effects of riparian forest buffers on aquatic ecosystems; 4) the time to recovery after harvest of trees or reestablishment of riparian buffer systems; and 5) the effects of underlying soil and geologic materials on chemical, hydrological, and biological processes.

This report examines the scientific basis for applying the existing RFBS specification as an agricultural Best Management Practice (BMP) in the different physiographic provinces of the Chesapeake Bay Watershed (CBW, Table 1 and Figure 1). The report briefly reviews NPS pollution problems in the Bay Watersheds and approaches to NPS pollution control (Sections I. B & C); the scientific foundation for the Riparian Forest Buffer System specification (Sections I. D & E); and the water quality functions of each of the three zones of the RFBS (Section I. F). Included is a review of the existing research on RFBS in different physiographic provinces that comprise the CBW (Section II). Based on these results, the effectiveness of RFBS for NPS pollution control is characterized in different parts of the Bay watershed (Section III). Finally, research needs are discussed in Section IV. RFBS are one of many factors that influence water quality and stream health. A complex suite of interrelated functions and mechanisms contribute to water quality and physical habitat parameters of the aquatic ecosystem. Other important factors, outside the scope of this report, that may affect the functioning of RFBS, and should be considered in their design, include: the type and intensity of land use in the watershed; the effectiveness of stormwater management; streambank and streambed stability; and stream uses (recreation, water supply, etc.).

B. NONPOINT SOURCE POLLUTION CONTROL RELATIVE TO NUTRIENT LOAD REDUCTION STRATEGIES

Nonpoint source pollution is the major cause of surface water impairment in the United States (Baker, 1992; Long, 1991) and has been addressed as a national priority since passage of the Clean Water Act (CWA), Section 319, which requires "that programs for the control of nonpoint sources of pollution be de-

TABLE 1
Land use in physiographic regions of Chesapeake Bay Watershed (NCRI Chesapeake, 1982).

Physiographic Region	Crop	Forest	Wetland	Other	TOTAL	% of TOTAL
	ha					
Appalachian Plateau	659,700	2,611,100	181,400	658,800	4,111,000	28
Valley & Ridge	986,200	2,659,600	60,500	911,100	4,617,400	32
Piedmont	825,500	1,607,900	141,300	688,100	3,262,900	23
Coastal Plain	768,600	1,020,400	509,300	119,800	2,418,000	17
TOTAL	3,240,000	7,899,000	892,500	2,377,800	14,409,300	100
% of TOTAL	22%	55%	6%	17%	100%	

Figure 1 shows generalized physiographic regions.

veloped and implemented." The effectiveness of the RFBS is likely to be judged by their NPS pollution control effectiveness.

Although assessments are incomplete and do not include all states, estimates are that about 30% of US waters are impaired—i.e. they do not fully support their designated uses (USEPA, 1990a). Of impaired waters, about two-thirds of the problems are primarily from NPS pollution (USEPA, 1986). The nonpoint sources of pollution vary, but agriculture is the major contributor for rivers and lakes. Besides agriculture, the other major contributors of NPS pollution are urban areas, mining, atmospheric deposition, and natural origins. Nutrients and sediments are still the principal sources of surface water impairment (USEPA, 1986; USEPA, 1990a; USEPA, 1990b). Sediments are the most important cause of impairment for rivers, and nutrients are the most important cause of impairment for estuaries. Pesticides, metals, and priority pollutants are identified as problems in less than 20% of the assessed waters. The extent of contamination, especially for pesticides, may be underestimated.

The earliest assessments of Chesapeake Bay water quality in the 1980's identified non-point source pollution as a major cause for water quality impairment in the Bay (Correll, 1987; Chesapeake Bay Program, 1991). Reduction of NPS pollution has been a significant part of the strategy to improve water quality in Chesapeake Bay since that time. The main problems were identified as nutrient enrichment, high levels of toxic substances, and excessive sediment loads. Effective control of all these types of pollution, especially nutrients and sediments, requires a watershed based program for NPS pollution control.

Improvement and maintenance of water quality is the single most important component of the overall protection and restoration plan established in the 1987 Chesapeake Bay Agreement (Chesapeake Bay Program, 1991). One of the most ambitious goals of the 1987 and 1992 agreements is to reduce nutrient loadings to the Bay by 40% by the year 2000 and to retain this level as a permanent cap on nutrient levels. Strategies for nutrient load reduction require control of both point and nonpoint sources of pollution. Based on 1985 land uses and results from a Watershed Model (Donigian et al., 1990), nonpoint sources dominate both N (53% of total) and P (68% of total) loads to the mainstem of the Bay (Chesapeake Bay Program, 1991, L. Shuyler, personal communication, 1995). The Watershed Model has been used to estimate edge-of-stream nitrogen and phosphorus loadings from various land uses in the Bay watersheds. Agriculture (including conventional cropland, conservation cropland, pasture, and animal waste facilities) accounted for 69% of total N and 79% of total P in NPS pollution.

Of the entire loadings of N and P to the mainstem of the Bay (point plus NPS), 44% of the N and 50% of the P came from agricultural nonpoint sources (L. Shuyler, personal communication, 1995). The

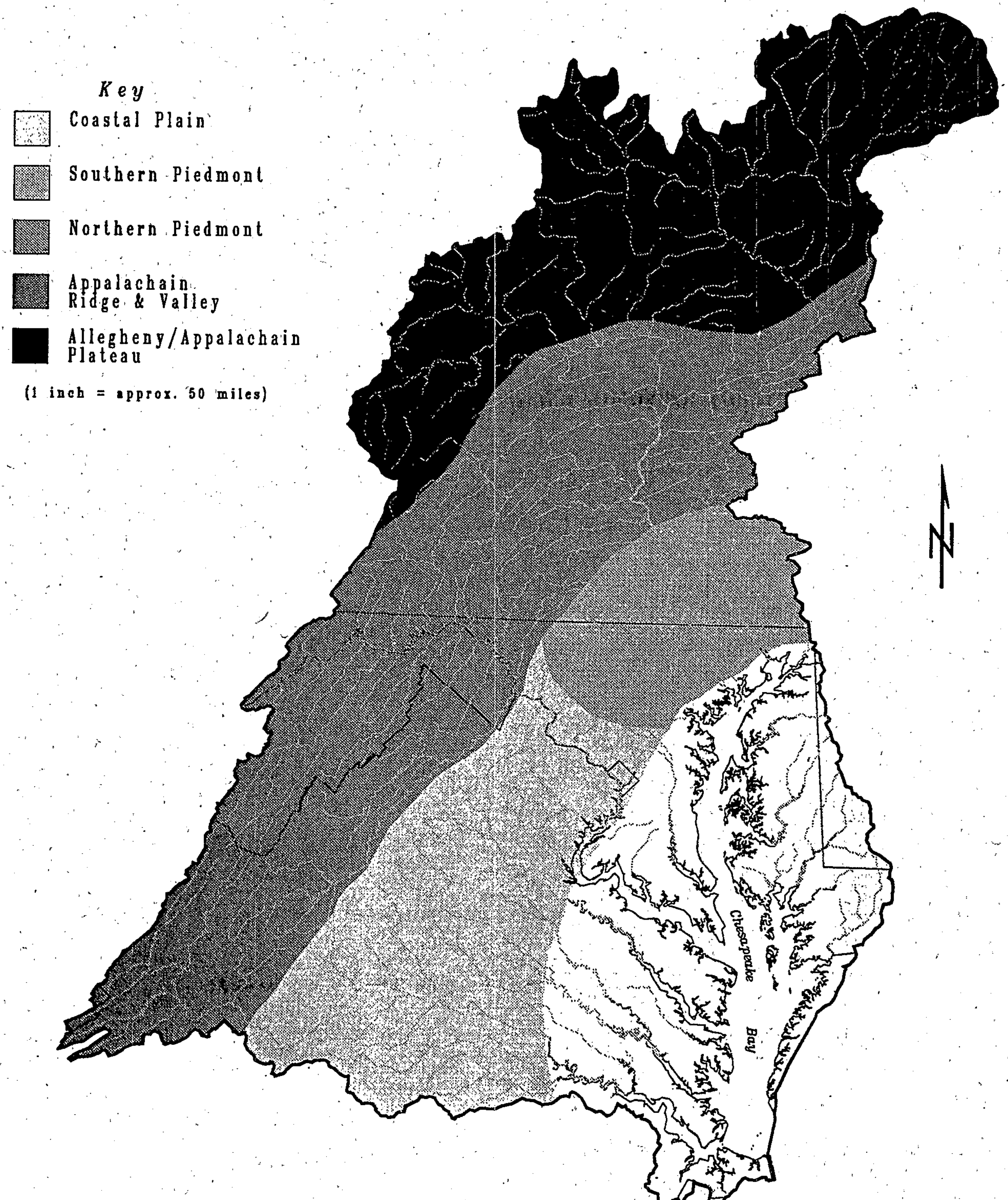

FIGURE 1. Physiographic regions of the Chesapeake Bay Watershed.

Susquehanna Basin (Pennsylvania, New York) and the Eastern Shore (Delaware, Maryland, Virginia) contribute the highest NPS loads of N and P. Loads from these two regions were dominated by agricultural sources. In the Susquehanna, 74% of NPS N loads were from agriculture. Nonpoint source loads of N from the Eastern Shore were 81% agriculturally related (Chesapeake Bay Program, 1991).

Until 1990, approaches for NPS pollution control in the Bay watersheds were largely focused on controlling upland sediment and sediment-borne pollutants (Chesapeake Bay Program, 1990). These traditional approaches were a combination of source reduction (i.e. reduce erosion rates in fields) and engineered buffer systems or structural BMPs such as ponds, sediment detention basins, terraces, grass water ways and vegetated filter strips. In 1990, the Chesapeake Bay Program's Nonpoint Source Evaluation Panel recommended a systems approach for nutrient load reduction with regional and watershed management strategies based on watershed mass-balances (Chesapeake Bay Program, 1990). A systems approach for NPS pollution reduction will include structural BMPs and source load reductions, as well as approaches which seek to integrate the management and restoration of landscape features which retain pollutants through a combination of ecosystem processes. Examples of these pollutant sinks include natural wetlands, constructed wetlands, and riparian forest buffer systems (Fields, 1992). As pollutant sinks increase in complexity from simple physical structures to diverse natural ecosystems, both the importance and difficulty of understanding processes which sequester or remove pollutants also increase.

C. WATERSHED APPROACHES TO NONPOINT POLLUTION ESTIMATION AND ABATEMENT

Risk assessment and source reduction are new approaches for NPS pollution control (Baker, 1992). A high percentage of total pollutant loadings in some watersheds comes from a relatively small portion of the watershed area because of improper management of sources, improper siting of facilities, problematic environmental and site conditions, or a combination of these factors. Watershed scale risk assessment seeks to identify and reduce loadings from areas which contribute large amounts of NPS pollution.

Concurrent with identification of problem areas comes the opportunity for source reduction. Source reduction has been responsible for some of the more impressive successes of NPS pollution reduction, including the reduction of loadings of lead from automobile emissions and of organo-chlorine pesticides (Baker, 1992). Source reduction should be linked with watershed-scale risk assessments because the potential for source reduction may be greatest (and probably most economical) in areas which are generating highest unit area loadings. The linkage of risk assessment and source reduction will depend on interacting factors such as type of pollutant (e.g. purchased input vs. by-product), reason for high risk (e.g. poor management, siting of facilities, inherent regional risks), and availability of alternative practices and/or sites.

Even when risk assessment and source reduction strategies lead to load reductions under average conditions, a third aspect of watershed management - maintenance and restoration of buffer systems between terrestrial and aquatic ecosystems - is necessary to reduce the contributions of extreme events to NPS pollutant loads. Under the best of conditions, source reduction will likely leave watersheds vulnerable to extreme events, including both weather extremes as well as economically generated extremes (e.g. intensification of pollution generating production practices). Watershed studies have demonstrated the importance of extreme events to water and pollutant transport. Extreme events within a year dominate annual totals and wet years within multi-year cycles dominate long-term loadings (Jaworski et al., 1992; Lowrance and Leonard, 1988; Magnien et al., 1992). Control of NPS pollution from extreme events will require integrating risk assessment and source reduction approaches with buffer systems as landscape scale "insurance policies."

Buffer systems are also important components of watershed NPS pollutant control efforts because of the limitations of other BMPs for NPS pollution control. For example, Hall (1992) monitored changes in groundwater nitrate (NO_3—N) concentrations beneath heavily fertilized and manured fields in Lancaster County, PA following the implementation of "input management" techniques. Fertilizer/manure inputs were decreased from 39 to 67% (222 to 423 kg ha^{-1}) but groundwater nitrate, changed by -12 to 50%. By the end of the study, nitrate concentrations in groundwater still exceeded federal drinking water standards. Shirmohammadi et al. (1991) used the CREAMS simulation model to evaluate the effects of seven different BMPs on groundwater nitrate concentrations beneath cropping systems on the eastern shore of Maryland. Although CREAMS does not provide absolute predictions, none of the BMPs were predicted to reduce groundwater nitrate concentrations to less than the federal drinking water standard. Under appropriate conditions, described in this report, RFBS are likely to be an important component of NPS pollution control when in-field BMPs are inadequate.

D. HISTORICAL OVERVIEW OF SCIENTIFIC INTEREST IN RIPARIAN ECOSYSTEMS

Most of the knowledge of riparian ecosystem effects on water quality comes from research conducted since 1975. Two publications in 1978 galvanized scientific and management interest in riparian ecosystems. Karr and Schlosser (1978) concluded that stream environments are largely controlled by adjacent riparian ecosystems and provided an overview of relationships between water resources and riparian ecosystems (the land-water interface). Johnson and McCormick (1978) edited the proceedings of a symposium which included 55 reports on various aspects of riparian research, management, and policy. While the symposium proceedings contained excellent discussions of the late 1970's state-of-knowledge concerning riparian ecosystems and other types of wetlands (Brown et al., 1978; Wharton and Brinson, 1978) only one paper (Mitsch, 1978) dealt specifically and quantitatively with the water quality functions of a riparian ecosystem. The proceedings also included a review of the general water quality functions of wetlands (Kibby 1978) in which a number of publications on nutrient cycling in riparian and other wetlands were cited. Only a few of the citations dealt specifically with water quality effects of riparian ecosystems (Kitchens et al., 1975; Lee et al., 1975; Kuenzler et al., 1977; Richardson et al., 1978). Although the 1978 symposium contained numerous claims about the water quality functions of riparian ecosystems, few data were presented.

In the late 1970's a number of research projects began to develop a more quantitative understanding of the role played by riparian ecosystems in controlling NPS pollution by sediment and nutrients in agricultural watersheds (Jacobs and Gilliam, 1985b, Lowrance et al., 1983; Peterjohn and Correll, 1984). These studies were primarily in the Coastal Plain physiographic province of the Eastern U.S., where the typical land-use pattern is intensive row-crop agriculture in upland areas with riparian forests along low-order streams. These early studies shared at least two other important characteristics: 1) a relatively shallow aquiclude which forced most infiltrated water to move laterally toward streams and pass through or near the riparian forest root zone and 2) naturally regenerated forests typical of the region rather than forests managed specifically for water quality functions. These studies focused on riparian processes related to nutrients and sediment with little or no attention to the fates of other pollutants or to the effects of riparian areas on the physical or trophic status of the stream.

As interest in the nonpoint pollution control value of riparian ecosystems increased, recognition of their importance to the physical and trophic status of streams also developed. Karr and Schlosser (1978) quantified the effects of riparian vegetation on sunlight penetration and temperature of streams. Research in the 1980s confirmed the importance of large woody debris and leaf litter inputs to the habitat and trophic status of most small streams (Meyer and O'Hop, 1983; Benke et al., 1985; Harmon et al., 1986). By 1987, it was well established that woody debris derived from riparian forests played an important role in controlling channel morphology, the storage and routing of organic matter and sediment, and the amount and quality of fish habitat (Bisson et al., 1987).

E. RESEARCH BACKGROUND FOR THE RIPARIAN FOREST BUFFER SYSTEM SPECIFICATION

By the late 1980s, there was a clear need to synthesize the existing knowledge into management recommendations for the establishment, maintenance, and management of riparian ecosystems for a broad range of water quality functions (Lowrance, 1991). In 1991, the United States Department of Agriculture-Forest Service (USDA-FS) with assistance from USDA-Agricultural Research Service, USDA-Soil Conservation Service, Stroud Water Research Center, PA, Pennsylvania Dept. of Environmental Resources, Maryland Dept. of Natural Resources, and U.S. Dept. of Interior Fish and Wildlife Service developed draft guidelines for riparian forest buffers. This effort resulted in a booklet entitled "Riparian Forest Buffers - Function and Design for Protection and Enhancement of Water Resources" (Welsch, 1991) which specified a riparian buffer system consisting of three zones (Figure 2).

Zone 1 is permanent woody vegetation immediately adjacent to the stream bank. Zone 2 is managed forest occupying a strip upslope from Zone 1. Zone 3

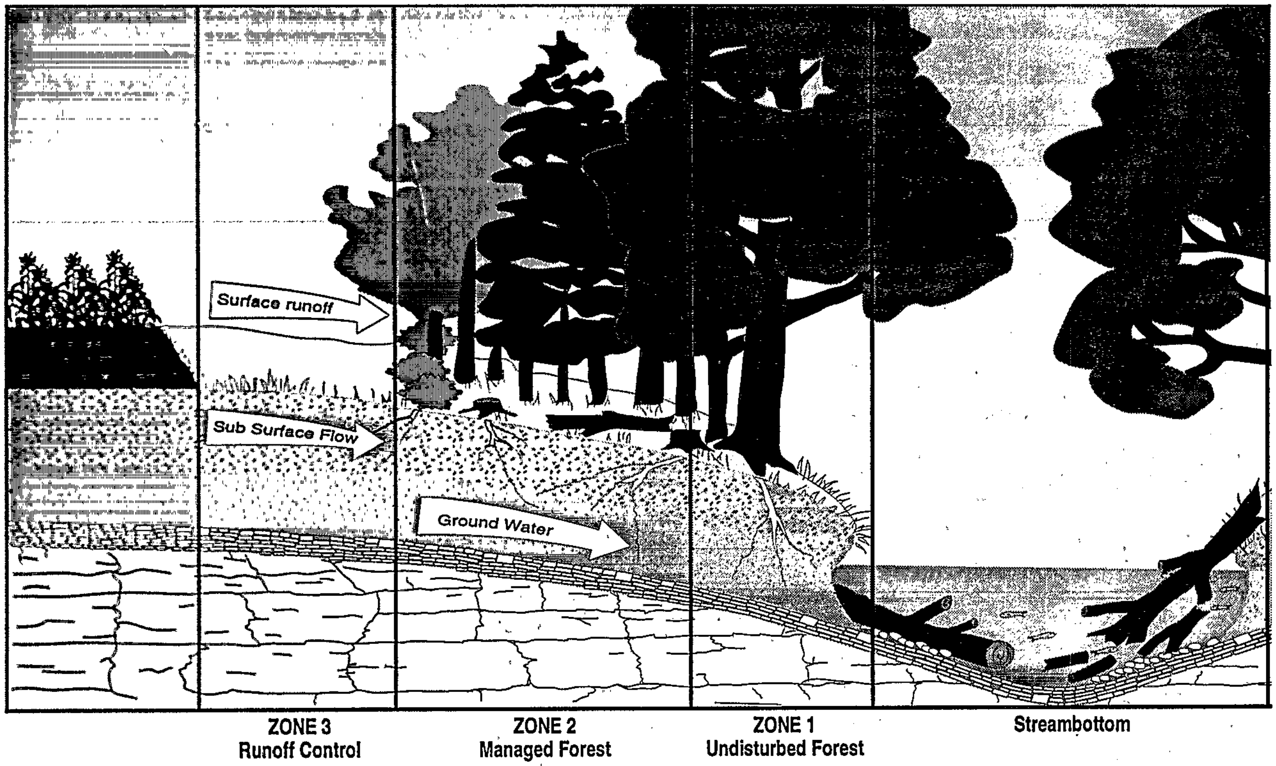

FIGURE 2. Schematic of the three zone Riparian Forest Buffer System.

is an herbaceous filter strip upslope from Zone 2. The specification applies to areas where cropland, grasslands, and/or pasture are adjacent to riparian areas on a) permanent or intermittent streams, b) margins of lakes and ponds, c) margins of wetlands, or d) margins of groundwater recharge areas such as sinkholes. Although referred to as a riparian forest buffer, inclusion of the non-forested herbaceous strip as Zone 3 suggests that a more correct name would be "Riparian Forest Buffer System".

The primary purposes of Zone 3 of the RFBS are to remove sediment from surface runoff and to convert channelized flow to sheet flow. The primary function of Zone 2 is to block transport of sediment and chemicals from upland areas into the adjacent wetland or aquatic ecosystem. Vegetation and litter in these zones forms a mechanical barrier to sediment transport. Plant roots take up chemicals that become sequestered in growing biomass. Vegetation also produces organic matter that fosters chemical and biological processes that immobilize or transform pollutants. Although most Zone 2 functions also occur in Zone 1, the primary purpose of Zone 1 is to maintain the integrity of the stream bank and a favorable habitat for aquatic organisms. Shade and litterfall provided by streamside vegetation has a direct influence on water temperature and dissolved chemicals.

The USDA-FS report and specification were based on a synthesis of literature existing through 1989 and on in-depth discussions with scientists and managers working on various riparian ecosystems (Welsch, 1991). Some of the generalizations which guided the design of the RFBS were based on studies of nutrient sequestering and nutrient transformations in agricultural watersheds (Correll, 1983; Lowrance et al., 1985; Yates and Sheridan, 1983). These watershed-scale studies indicated that riparian forests were important nutrient and sediment sinks in agricultural watersheds, but provided little or no guidance on how to design an effective RFBS. Process studies in these and other systems provided most of the original design guidance. Several studies on nitrate removal from shallow groundwater in riparian forest buffers found that most reduction in nitrate concentration takes place within the first 10 to 15 m of forest (Lowrance et al., 1984a, Peterjohn and Correll, 1984, Jacobs and Gilliam, 1985b) and that the necessary width for shallow groundwater nitrate removal could be relatively short. Although effective in reducing sediment and sediment borne chemical concentrations in sheet flow (Peterjohn and Correll, 1984), it was known that channelized flow can bypass riparian forests. To control channelized flow into a riparian forest, a herbaceous strip in Zone 3 could be much more easily reshaped and revegetated than a forest. Herbaceous buffers, especially grass filters, are effective at removing coarse suspended sediments and some sediment-borne pollutants but may require frequent maintenance and are not very effective at nutrient removal from shallow groundwater (Dillaha et al., 1989; Magette et al., 1987; Magette et al. 1989).

Long-term sequestering and removal of nutrients and other contaminants in the RFBS is the main purpose of Zones 3 and 2. This can occur by 1) accumulating sediment and adsorbed contaminants; 2) microbial transformations (for N) and biochemical degradation (for pesticides); and 3) incorporation of nutrients and other chemicals into woody biomass and soil organic matter. At least one study of Coastal Plain riparian forests showed substantial amounts of nutrient sequestering in woody biomass (Fail et al., 1986). The RFBS specification encourages production and harvest of woody biomass from Zone 2 to remove nutrients and other contaminants. Once vegetation has been removed from the stream channel, recovery through plant succession may take long periods of time and revegetation may be dominated by undesirable species (Sweeney, 1993). Therefore, the need for permanent control of the stream physical and trophic environment requires directed succession toward desirable permanent vegetation in those portions of the RFBS which directly influence the stream channel, in particular Zone 1.

A number of practical concerns were also considered in the RFBS specification (Welsch, 1991). Most of the RFBS should be available for management to provide an economic return without sacrificing water quality functions. Characteristics of soils, hydrology, and potential vegetation should guide design and planning of effective RFBS. The RFBS should be used in conjunction with sound upland management practices including nutrient management and erosion control. In-stream woody debris removal should be limited, but woody debris with potential to form dams which cause inundation should be removed. The dimensions of the RFBS should depend on the existing and potential NPS pollutant loads and the minimum size for sustained support of the aquatic environment.

F. CURRENT UNDERSTANDING OF RFBS FUNCTIONS

Several of studies are underway to test the effectiveness of RFBS which correspond to or are similar to the USDA specification. Vellidis et al. (1993) and Sweeney (1993) describe RFBS restoration projects in the Georgia Coastal Plain and the Pennsylvania Piedmont, respectively. Beare et al. (1994) describe preliminary results from management of an existing riparian forest which involves establishment of Zone 3 adjacent to mature riparian forest and tree harvest treatments in Zone 2. Schultz et al. (1994) describe a multi-species three zone buffer system for use in agricultural areas of Iowa and other parts of the Midwest. Much of the current understanding of RFBS has been incorporated into a Riparian Ecosystem Management Model which simulates hydrologic and nutrient cycling processes in RFBS that conform to the USDA specification (Altier et al., 1994; Sheridan et al., 1993).

It is important to note that our current understanding of the functions of the RFBS is based on studies that have been done in areas where riparian forests currently exist due to a combination of hydrology, soils, cultural practices, and economics. Most of our current knowledge of the functions of the three zones of the RFBS specification is derived from studies in existing riparian forests and on experimental and real-world grass buffer systems. Although results can be extrapolated from these existing forests to restored RFBS, most of the study sites are actually at some stage of restoration, following clearing within the last 20-80 years.

1. Zone 1—Control of the Stream Environment

Although reduction of NPS pollution is a widely recognized function of RFBS, they also contribute significantly to other aspects of water quality and physical habitat (Allan and Flecker, 1993; Karr, 1993). Habitat alterations, especially channel straightening and removal of riparian vegetation, continue to impair the ecological health of streams more often and for longer time periods than toxic chemicals (Hughes et al., 1990). Sweeney (1992) considers loss of riparian forests in eastern North America to be one of the major causes of aquatic ecosystem degradation.

Zone 1, the permanent woody vegetation at the stream edge, enhances ecosystem stability and helps control the physical, chemical, and trophic status of the stream. Healthy riparian vegetation in Zone 1 also contributes to bank stability and minimizes instream sediment loading due to bank erosion. Zone 1 also has substantial ability to control NPS pollution through denitrification (Ambus and Lowrance, 1991; Lowrance, 1992; Schnabel, 1986), sedimentation (Lowrance et al., 1986), or direct root uptake of pollutants.

Zone 1 vegetation controls light quantity and quality, moderates temperature, stabilizes channel geometry, provides tree roots and woody debris for habitat, and provides litter for detritivores (Barton et al., 1985; Beschta et al., 1987; Hax and Golladay, 1993; Hill and Harvey, 1990; Karr and Schlosser, 1978; Sweeney, 1992, 1993). To maintain the biological integrity of the aquatic ecosystem, an ideal managed buffer system should have patterns of vegetation, litterfall, and light penetration similar to those in a natural, undisturbed riparian forest (Golladay and Webster, 1988; Karr, 1993; Sweeney, 1992, 1993). However, for many locations, representative sites of truly natural, undisturbed riparian ecosystems do not exist. In fact, after a long history of human disturbance in many areas, the concept can be difficult to define (Bren, 1993). Karr (1993) suggests that within a homogeneous region, relatively pristine areas may be identified as benchmarks for the evaluation of other sites.

Riparian forest buffer functions related to protection of the stream environment will not be reviewed for different physiographic regions because there is general agreement among literature sources on the need for riparian forests in the Eastern U.S. for this purpose. The major differences among physiographic regions appear to be in the importance of stream temperature control for cold-water vs. warm-water fisheries.

a. Temperature and Light

The diel and seasonal patterns of water temperature are critical habitat features that directly and indirectly affect the ability of a given stream to maintain viable populations of most aquatic species, both plant and animal. Considerable indirect evidence suggests that the absence of riparian forests along many streams and rivers in the Chesapeake drainage, particularly in agricultural areas, may have a profound effect on the current geographic distribution of many species of macroinvertebrates and fish. Sweeney (1992) reviewed the effects of temperature alterations on the growth, development, and survival of stream macroinvertebrates found in the Pennsylvania Piedmont. These studies showed that temperature changes of 2–6°C usually alter key life-history characteristics of most of the study species

In the absence of shading by a forest canopy, direct sunlight can warm stream temperatures significantly, especially during summer periods of low flow. For example, maximum summer temperatures have been reported to increase 6–15°C following deforestation (Beschta and Taylor, 1988, Lee and Samuel, 1976, Brown and Krygier, 1970). Streams flowing through forests will warm very rapidly as they enter deforested areas, but excess heat dissipates quickly when streams reenter the forest. Burton and Likens (1973) demonstrated this alternate warming (by 4 to 5°C) and cooling as a stream passed through clear-cut and uncut strips in the Hubbard Brook Experimental Forest, New Hampshire. In Pennsylvania (Valley and Ridge Province), average daily stream temperatures that increased 11.7°C through a clearcut area, were substantially moderated after flow through 500 m of forest below the clearcut. The temperature reduction was attributed primarily to inflows of cooler groundwater (Lynch et al., 1980). The impact of deforestation on stream temperature varies seasonally. In the Pennsylvania Piedmont, Sweeney (1993) found that from April through October average daily temperatures in a second-order meadow stream reach were higher than in a comparable wooded reach, but that the reverse was true from November through March.

Riparian forest buffers have been shown to prevent the disruption of natural temperature patterns as well as to mitigate the increases in temperature following deforestation (Brown and Krygier, 1970; Brazier and Brown, 1973; Lee and Samuel, 1976). Brazier and Brown (1973) found that buffer strips of 10 m width were as effective as a complete forest canopy in reducing solar radiation reaching small streams in the Pacific Northwest. The exact width of Zone 1 needed for temperature control will vary from site-to-site depending on a variety of factors. Brown (1974) pointed out that streams oriented in a north-south direction are less easily shaded than streams flowing east or west, and that a buffer on the north side of a stream may have little or no effect. Also, in larger streams and rivers, the width of the channel prevents a complete canopy cover, so that the effect of canopy shading may be reduced. In eastern North America, openings in the canopy immediately above streams occur when the channel width exceeds about 20 m in width (i.e., about stream order 4 or 5). In a study of five Minnesota Rivers, Sinokrot and Stefan (1993) inferred midsummer shading of 40-60% for rivers ranging from 15-50 m in width but effectively no shading along extremely wide rivers (e.g., the 300 m wide Mississippi R.). Stream orientation relative to solar angle may also affect the extent of shading for larger streams. Although shading on larger rivers may have little or no effect on water temperature, shaded stream banks provide habitat microsites for fish and other aquatic organisms.

The ability of a given width of streamside forest to maintain or restore the natural temperature characteristics of a stream segment depends on how it affects the factors that control the diel and seasonal thermal regime of the stream. Such factors (other than shading) include: flow, channel geometry, solar radiation, evaporative heat loss, conductive surface heat exchange, and, in some cases, conductive heat exchange with the streambed. Heat budget models can integrate local meteorological data with the above factors to predict stream and river temperatures with relatively high precision (e.g., Edinger et al., 1968; Brown, 1969; Beschta, 1984; Theuer et al., 1984; Sinokrot and Stefan, 1993; Edinger and Buchak, in press). These models indicate that solar radiation is the major factor influencing peak summer water temperatures and confirm that shading by the streamside forest is critical to the overall temperature regime of a stream or river. Stefan and Sinokrot (1993) estimated that removal of the forest canopy along the Straight R., Minnesota, would increase average summer water temperatures approximately 6 C.

Hewlett and Fortson (1982) measured unexpectedly large stream temperature fluctuations in the Georgia Piedmont on a clearcut site with a 5 to 8 m buffer strip left on each side of a first-order stream. After logging and wind damage, about a 50% cover canopy remained over the stream. Despite the partial buffer, as well as rapid regrowth of low vegetation over the stream, stream temperature fluctuations for four years following logging were much greater than in an uncut forest. Since the measured temperatures could not be accounted for by a stream temperature model, the authors suggested that in addition to the effects of direct radiation on stream temperature, effluent groundwater temperatures may also have been modified by the removal of vegetative cover.

b. Habitat Diversity and Channel Morphology

The biological diversity of streams depends on the diversity of habitats available. Woody debris is one of the major factors in habitat diversity. Woody debris can benefit a stream in several ways: (1) by stabilizing the stream environment through attenuation of the erosive influence of stream flow; (2) by increasing the diversity and amount of habitat for aquatic organisms; (3) by providing a source of slowly decomposable nutrients; and (4) by forming debris dams, it enhances the availability of nutrients for aquatic organisms from more rapidly decaying material.

The quantity of woody debris in streams under forested canopies in the Eastern United States has been reported to range from 0.4 to 23 kg m^{-2}, averaging about 8 kg m^{-2} (Webster et al., in press). These figures are undoubtedly lower than would be encountered in streams flowing through undisturbed forest because most eastern streams have been subjected to extensive removal of streamside vegetation and, in larger streams, clearing of woody debris for navigational purposes (Webster et al., in press). Quantities of large woody debris (LWD) recommended for healthy streams in the George Washington National Forest in Virginia range from 34 pieces of LWD per km for warm water fisheries to 136 pieces/km for cold water fisheries. Although the quantity of woody debris in streams without forested riparian zones would be expected to be very low, there are few quantitative studies. Sweeney (1992) found that the volume of woody debris under forested canopies in a Mid-Atlantic Piedmont stream was 20 times greater than the volume in a comparable meadow reach. Following removal of a riparian forest, LWD present in the stream declines through gradual decomposition, flushing during storms, and lack of inputs. Smaller debris from second-growth stands promotes less stability of the aquatic habitat and tends to have a shorter residence time in the stream.

Loss of streamside forest can lead to loss of habitat through stream widening where no permanent vegetation replaces forest or through stream narrowing where forest is replaced by permanent sod. In the absence of other perennial vegetation, bank erosion and channel straightening can occur as unimpeded streamflow scours the streambed and banks (Hartman et al., 1987; Oliver and Hinckley, 1987). The accelerated streamflow velocity allowed by straight channels promotes channel incision as erosion from the stream bottom exceeds sediment entering the stream. This process can eventually lead to the development of wide, shallow streams that support an impoverished diversity of species (Shields et al., 1994). Bisson et al. (1987) point out that stability of debris accumulation is important for aquatic habitat. Because of the greater resistance to displacement by hydraulic forces, LWD is of greater benefit to stream stability. Longer material is relatively more important for the stability of wider streams.

In contrast, narrowing of stream channels has also been reported following the replacement of streamside forest with permanent grassland or grass sod. Zimmerman et al. (1967) found that the narrowing of deforested stream channels was evident for streams up to drainage areas of 13 km^2 (5 mi^2) or about a third or fourth-order stream. Sweeney (1992), quantified the narrowing phenomenon more explicitly in a Pennsylvania Piedmont basin, showing that: (1) first and second-order wooded reaches averaged about 2 times wider than their meadow counterparts of the same order; and (2) third and fourth-order forested reaches were about 1.7 times wider than in deforested areas. The channel narrows in the absence of a streamside forest because grassy vegetation, which is normally shaded out, develops a sod that gradually encroaches on the channel banks. For benthic macroinvertebrates, microbes, and algae, which live in and on the substratum, the loss in stream width translates into a proportionate loss of habitat. The effects of channel narrowing on fish habitat are more complex and involve the influence of woody debris on the pool and riffle structure (as discussed below).

Links between LWD in streams, the abundance of fish habitat, and the populations, growth, and diversity of fishes have been documented (see reviews by Dolloff, 1994; Harmon et al., 1986; Bisson et al., 1987). Even when selective harvesting of trees has been allowed along streams, the removal of old growth has caused a decline in aquatic habitat quality due to diminished inputs of LWD (Bisson et al., 1987). The surfaces of submerged logs and roots provide habitat that often support macroinvertebrate densities far higher than on the stream bottom itself (Rhodes and Hubert, 1991; Sweeney, 1992; Benke et al., 1984).

Woody debris, like boulders and bedrock protrusions, tends to form pools in streams either by directly damming flow, by the scouring effects of plunge pools downstream of fallen logs, or by forming backwater eddies where logs divert flow laterally (Dolloff, 1994a). In undisturbed forests, LWD can account for the majority of pool formation (Harmon et al., 1986; Hedman, 1992). As expected, removal of woody debris by deforestation typically results in loss of pool habitat (Bilby, 1984). Although pools are spatially contiguous with riffles, there is little or no overlap in the species composition of the dominant macroinvertebrates occurring in the two habitats. The loss of pools, therefore, translates directly into lower populations and diversity for this group. For fish, pools improve habitat by providing space, cover, and a diversity of microenvironments. Greater depth and slower velocity in pools afford protection to fish during storms, drought, etc. (Dolloff, 1994a). The habitat provided by LWD may also offset the destruction of stream habitat structures such as pools, riffles, and cascades by catastrophic storm events (Dolloff et al., 1994b).

Debris dams of large woody material block the transport of both sediment and smaller litter materials. The impoundment and delayed transport of organic material downstream enhances its utilization by aquatic organisms. By slowing transport rates, dams on small order streams serve as buffers against the sudden deposition of sediment downstream Bisson et al. (1987). The capacity of a stream to retain debris, therefore, is an important characteristic influencing the aquatic habitat. (Bisson et al., 1987; Meehan et al., 1977).

Although it is often thought that LWD is less important on large rivers and openwater habitats, it has been shown that woody debris derived from riparian forests along tidal shorelines of the Bay provides an important refuge habitat for numerous species of fish and crustaceans (Everett and Ruiz, 1993). Shallow water habitats with abundant LWD support greater abundances of many species of fish and crustaceans than do areas with no woody debris bordered by narrow strips of march (Everett and Ruiz, 1993; Ruiz et al., 1993). They hypothesize that the importance of LWD along Bay shorelines has been increased due to loss of habitat in submerged aquatic vegetation and oysterbeds.

c. Food Webs and Species Diversity

The two primary sources of food energy input to streams are litterfall (leaves, twigs, fruit seeds, etc.) from streamside vegetation and algal production within the stream. Total annual food energy inputs (litter plus algal production) are similar under shaded and open canopies, but the presence or absence of a tree canopy has a major influence on the balance between litter input and primary production of algae in the stream.

Meehan et al. (1987) noted that "streams flowing through older, stratified forests receive the greatest variation in quality of food for detritus-processing organisms." In the Piedmont, streams flowing through forested landscapes do not subsidize downstream channels that have been deforested (even contiguous reaches) because the large pieces of litter do not move very far (Sweeney, 1992). This means that a streamside forest is needed along the entire length of a stream in order to assure a proper balance of food inputs appropriate to the food chain of native species. Macroinvertebrate populations are affected by changes in litter inputs. The activity of benthic organisms may increase following streamside plant removal. Woody material decomposes more quickly following riparian forest removal, thereby further reducing the stream's nutrient retention (Golladay and Webster, 1988).

The quantity and quality of algal production in a stream is greatly affected by the quantity and quality of light striking its surface. For example, Bilby and Bisson (1992) showed that the algal community of a stream heavily shaded by an old growth forest was dominated by diatoms all year, while a nearby stream in a deforested area contained mainly filamentous green algae in the spring and diatoms at other times. Other studies have also shown that deforested sites tend to be dominated by filamentous algae while diatoms prevail under dense canopy cover (Lowe et al., 1986; Feminella et al., 1989). In the eastern Piedmont, filamentous algae such as *Cladophora* can be dominant in deforested streams due primarily to the a combination of high nutrients, high light levels, and warm temperature. Although some macroinvertebrates such as crayfish (Feminella and Resh, 1989) and waterboatmen insects (Sweeney and Schnack, 1976) readily consume this type of algae, most herbivorous species of stream macroinvertebrates have evolved mouthparts specialized for scraping diatoms from the surface of benthic substrates (Merritt and Cummins, 1984) and cannot eat filamentous algae.

The influence of differences in the quality of algal production on the aquatic ecosystem is complex. Algal grazing species generally benefit from an increase in algal growth (Wallace and Gurtz, 1986; Perrin et al., 1987; Bilby and Bisson, 1992; Sweeney, 1992). Because the growth efficiency of insects is often higher on algae than on detritus, the opening of the canopy may increase the production of macroinvertebrates in these reaches. For example, Behmer and Hawkins, et al. (1986) found both higher biomass and densities for most grazer species in deforested sites relative to forested sites. The pattern is not clear, however, because Hawkins (1982) found higher biomass but lower densities of grazers in deforested versus forested sites. Newbold et al. (1980) observed in California streams that the benthic community in logged watersheds became dominated by a few algal feeding species. The diversity of the macroinvertebrate community was significantly lower than in unlogged watersheds, except where the stream was protected by a riparian buffer of 30 m or more. For buffer strips less than 30 m in width, the Shannon diversity was significantly correlated with buffer width.

2. Zone 2—Removal of Nonpoint Source Pollutants

The primary function of Zone 2 is to remove, sequester, or transform nutrients, sediments, and other pollutants. Because of its proximity to Zone 1, Zone 2 might also have direct impacts on the stream channel system and contribute to Zone 1 functions. The pollutant removal function of a Riparian Forest Buffer System depends on two key factors; 1) the capability of a particular area to intercept surface and/or groundwater-borne pollutants and 2) the activity of specific pollutant removal processes. Focusing on these two factors as regulators of buffer zone effectiveness is useful for evaluating the importance of a particular site as a buffer and for evaluating the three zone RFBS specification. In the sections below we review the major pollutant removal processes that operate in Zone 2 and discuss how these processes interact with pollutants in either surface runoff or groundwater flow in the context of the three zone specification.

a. Nitrate Removal

Nitrate removal from shallow groundwater has been the focus of many completed and ongoing studies. At least four separate studies at different sites in the Gulf-Atlantic Coastal Plain Physiographic Province have shown that concentrations of nitrate in shallow subsurface flow are markedly reduced after passage through portions of natural riparian forest analogous to Zone 2 (Jacobs and Gilliam, 1985a,b; Jordan et al., 1993; Lowrance et al., 1983, 1984a; Peterjohn and Correll, 1984). Studies in other physiographic settings have also shown nitrate removal from shallow groundwater in areas analogous to Zone 2 (Groffman et al., 1992; Simmons et al., 1992). Most studies with high levels of nitrate removal were in areas with high water tables that caused shallow groundwater to flow through or near the root zone.

The mechanisms for removal of nitrate in these study areas are thought to be a combination of denitrification and plant uptake. Linkages between plant uptake and denitrification in surface soils have been postulated as a means for maintaining high denitrification rates in riparian ecosystems (Groffman et al., 1992; Lowrance, 1992). In contrast, riparian systems without substantial contact between the biologically active soil layers and groundwater or with very rapid groundwater movement appear to allow passage of nitrate with only minor reductions in concentration and load. Correll et al. (1994) reported both high nitrate concentrations and high nitrate removal rates beneath a riparian forest where very high nitrate flux and rapid groundwater movement through sandy aquifer material limited nitrate removal efficiency. Staver and Brinsfield (1990) showed that groundwater flow beneath the biologically active zone of a narrow riparian buffer along a tidal embayment in Maryland resulted in little removal of nitrate. It is also known that groundwater discharging through sediments of tidal creeks may have up to 20 times the nitrate concentrations found in the main stem of the creeks (Reay et al., 1992).

Phillips et al. (1993) indicated that groundwater nitrate might bypass narrow areas of riparian forest wetland and discharge into stream channels relatively unaltered when the forest is underlain by an oxygenated aquifer. This pattern of groundwater flow was supported by modelling of a small Coastal Plain watershed in Maryland (Reilly et al., 1994). Isotopic analysis of groundwater and surface water in this watershed suggested that denitrification was not affecting the nitrate concentrations of discharging groundwater. In these cases where nitrate enriched water surfaces in the stream channel, a wide RFBS would have little effect on nitrate. Deeply rooted vegetation near the stream might have some effect.

Studies in New Zealand have shown that the majority of nitrate removal in a pasture watershed took place in organic riparian soils which received large amounts of nitrate laden groundwater (Cooper, 1990). The location of the high organic soils at the base of hollows caused a high proportion of groundwater (37-81%) to flow through the organic soils although they occupied only 12% of the riparian zone. A related study in New Zealand (Schipper et al., 1993) found very high nitrate removal in the organic riparian soils but streamflow was still enriched with nitrate. The authors speculated that water movement through mineral soils was responsible for most of the nitrate transport to streams. Riparian systems with intermingling of organic and mineral soils point out the need to understand where groundwater is moving and what types of soils it will contact, especially in seepage areas.

b. Plant Uptake

Maintenance of active nutrient uptake by vegetation in Zone 2 should increase the potential for short-term (non-woody biomass) or long-term (woody biomass) sequestering of nutrients. Although plant water uptake is chiefly a passive transpiration process, plant nutrient uptake is mostly an active process, dependent upon plant metabolic activity (Hoagland and Broyer, 1936). Most nutrients are transported into plants against an electrochemical potential gradient (Bowling et al., 1966; Higinbotham et al., 1967). Observations of ion concentrations in plant xylem exceeding external soil water concentrations by over 100 times indicate significant active uptake of P (Russell and Shorrucks, 1959). Transpiration tends to influence the uptake of a nutrient when the external concentration of that nutrient is high. Transpiration in riparian forests is very high and can control water movement to streams (Correll and Weller, 1989; Bosch et al., 1993). Kramer and Kozlowski (1979) pointed out that transpiration increases the mass flow of solutes toward root surfaces.

Nutrient uptake by flood-intolerant plants is strongly influenced by the aeration status of the soil (Hoagland and Broyer, 1936; Hopkins, 1956; Hopkins et al. 1950). As low oxygen supply decreases root metabolism, the uptake of most nutrients decreases. Flood-tolerant species, such as those found in many riparian forests, may tolerate low-oxygen conditions by means of adaptive metabolic responses (Crawford, 1982). They may also avoid root anoxia by morphological adaptations that facilitate the availability of oxygen. Under flooded conditions, roots may become thicker and increase in porosity, allowing an internal downward diffusion of oxygen (Armstrong, 1968; Coutts and Philipson, 1978). The growth of adventitious roots may also allow water and nutrient uptake from near-surface areas that are more aerated (Kozlowski, 1984; Sena Gomes and Kozlowski, 1980).

Vegetation selection for restored or managed RFBS must consider the ability of different species to take up and store nutrients under specific conditions of the site. Kozlowski and Pallardy (1984) point out that flooding can enhance the nutrient uptake and growth of some species. Bottomland hardwood seedlings grow faster under saturated conditions than under drained but well-watered conditions. More rapid increases in total dry weight and N and P uptake were found in water tupelo (*Nyssa aquatica* L.) as well as several other species under saturated conditions (Hosner and Leaf, 1962; Hosner et al., 1965). Shoot weights of a majority of wetland and intermediate plant species were either unaffected or increased under flooded conditions (Justin and Armstrong, 1987).

Nutrient uptake and accretion in riparian forests will be affected by vegetation management. Nutrient demand by vegetation corresponds with growth rate (Cole, 1981; McDonald et al., 1991). Loblolly pine dominated forests in the Gulf Coastal Plain attain maximum rates of growth of about 8t ha^{-1} yr^{-1} during the first twenty years of age, for which 101 kg of N and 9 kg of P are required each year (Nelson et al., 1970; Switzer et al., 1979). Cole and Rapp (1980) suggested a worldwide average annual N uptake rate of 70.5 kg ha^{-1} for deciduous tree species and 39 kg ha^{-1} for coniferous species. Temperate deciduous species produce 179 kg biomass kg^{-1} N uptake, and temperate coniferous species produce 103 kg biomass kg^{-1} N uptake (Cole and Rapp, 1980). However, Miller (1984) disputes the notion that coniferous forests require less nutrients than broad-leaved forests. His review of nutrient uptake studies indicates that the ranges of measured uptake for coniferous and broad-leaved forests overlap.

Compared to the "natural" riparian forests studied in most existing research, managed riparian forests have the potential for increased accumulation of N and P in biomass through both increased biomass production and increased foliar nutrient contents. Trees can respond to N subsidy by both increased growth rates and luxury N uptake. The growth rate of forests is commonly N limited. Cole (1986) suggested that high efficiency of N use by forests is an adaptation to the N-deficient environments that they frequently inhabit.

Often the potential N uptake rate is much higher than observed rates. Forest growth has been found to respond readily to N applications (Miller and Tarrant, 1983; Schmidtling, 1973). Mitchell and Chandler (1939) found large tree-growth responses to N fertilizer applications up to 400 to 600 kg ha^{-1}. Cole (1981) found that after fertilizing with 400 kg N ha^{-1} yr^{-1} in effluent from a municipal sewage treatment plant for three years, poplar (*Populus nigra* var. *italica* Muench.) and Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco) took up 213 and 78 kg N ha^{-1} yr^{-1}, respectively. This contrasted with an uptake of 16 kg N ha^{-1} yr^{-1} by poplar and 23 kg N ha^{-1} yr^{-1} by Douglas fir in unfertilized sites. Miller and Cooper (1973) showed that trees can take up "luxury" levels of N. Growth responses by 36-year-old Corsican pine (*Pinus nigra* var. *maritima* (Ait.) Melv.) to different levels of N fertilization showed that foliar N content reached a maximum of 26,400 mg kg^{-1} after applying the highest rate of 504 kg N ha^{-1} yr^{-1} for three years. Maximum volume growth corresponded to a foliar content of about 20,000 mg kg^{-1}, attained by applications of 336 kg N ha^{-1} yr^{-1} for three years.

Conditions do exist where N is no longer the limiting nutrient for forest growth. Long-term inputs of nitrogen, such as may occur from atmospheric deposition in the northeastern U.S., could result in N levels exceeding the total combined plant and microbial nutritional demands (Aber et al., 1989). Under these conditions, P might become the limiting factor for tree growth. Unlike upland forests, P may often be the most limiting nutrient in wetland ecosystems (Taylor et al., 1990). Mitsch et al. (1979) found the growth of bald cypress (*Taxodium distichum* (L). Rich.) in a southern Illinois swamp to correspond well with P inputs from flooding. Foliar P content of loblolly pine on wet Coastal Plain sites in South Carolina has been observed to correlate well with growth (Wells and Crutchfield, 1969). Analysis by Brinson (1977) of nutrient ratios in decaying litter from tupelo gum trees in a North Carolina swamp forest suggested that P levels may limit decomposition rates. If P is the limiting nutrient for tree growth, it should make vegetation an effective P sink.

While several studies have found plant uptake to be an important nutrient removal mechanism in areas analogous to Zone 2 of riparian forest buffers (Correll and Weller, 1989; Fail et al., 1986; Peterjohn and Correll 1984; Groffman et al. 1992), several factors may reduce the importance of plants as nutrient sinks. Pollutants in groundwater flowing into the riparian buffer will only be accessible to plants if the water table is high in the soil profile (Ehrenfeld 1987) or if mass movement of water due to transpiration demands moves water and solutes into the root zone. Coastal Plain riparian forests have been shown to control localized downslope water transport by creating moisture gradients which move water in unsaturated flow from both the adjacent stream and the upland field (Bosch et al., 1993). Nutrients in surface runoff and in water percolating rapidly through soil macropores as "gravitational water" may not be available to plants. Large rainfall events, that often transport a high percentage of pollutants in the CBW (Jaworski et al., 1992) often produce concentrated surface flow and macropore-dominated percolation.

Plant sequestering of nutrients is also limited by seasonal factors. In the temperate deciduous ecosystems that dominate riparian forest buffers in the CBW, plant uptake will decline or stop during the winter season. A high percentage of surface and groundwater flow occurs in the CBW during winter. There is also concern that nutrients trapped in plant tissues can be released back into the soil solution following litterfall and decomposition. However, nutrients released from decomposing plant litter may be subject to microbial, physical or chemical attenuation mechanisms in the root zone of forest soils. Storage of nutrients in woody tissue is a relatively long-term attenuation, but still does not result in removal of pollutants from the ecosystem unless biomass is removed. A final concern about plant uptake as a nutrient removal mechanism arises from the possibility that the ability of trees in a buffer zone to sequester nutrients in woody biomass becomes less as trees mature. The average tree age in most riparian forest buffers in the CBW is less than 100 years and should thus be accumulating nutrients in woody biomass. Although net vegetation accumulation of nutrients may reach zero, net ecosystem accumulation may continue as nutrients are stored in soil organic matter. Groffman et al (1992) describes a nitrate-enriched riparian system with symptoms of N saturation (Aber et al., 1989). Nitrogen saturation is not likely to occur in RFBS because of high denitrification rates removing N from the system.

Little is known about the types of vegetation needed in new or reestablished RFBS. Crop tree management (the selection and release of desired trees by removal of competing trees) will be possible in many natural successional riparian forests. Numerous native tree species are recommended for water quality improvement in crop tree management (Sykes et al., 1994). The trees were selected based on their ability as nutrient filters although little data exist on individual riparian species.

c. Microbial Processes

In addition to plant uptake, there are microbial processes that attenuate pollutants in RFBS. These processes include immobilization of nutrients, denitrification of nitrate and degradation of organic pollutants. Microbes take up or "immobilize" dissolved nutrients just as plants do. These immobilized nutrients can be re-released or "mineralized" following death and decomposition of microbial cells, just as nutrients sequestered by plants can be released following litterfall. In ecosystems that are accumulating soil organic matter, there will be a net storage of immobilized nutrients. Zone 2, if managed to foster soil organic matter accumulation, may thus support significant long-term rates of nutrient storage by immobilization.

Denitrification refers to the anaerobic microbial conversion of nitrate to N gases. Denitrification is controlled by the availability of oxygen (O_2), nitrate, and carbon (C). Although essentially an anaerobic process, denitrification can occur in well drained soils because of the presence of anaerobic microsites, often associated with decomposing organic matter fragments which deplete available oxygen (Parkin, 1987). It is likely that soil moisture gradients in riparian ecosystems cause a change in controlling factors within most three zone RFBS. In parts of the RFBS with better internal drainage and generally lower soil moisture conditions, denitrification may be generally limited by their interacting factors of carbon availability and aeration status. While many wetlands are often assumed to have high levels of denitrification because of high carbon soils and anaerobic conditions, denitrification in many wetlands will be N limited (Groffman, 1994). In the more poorly drained or wetland portions of a RFBS, denitrification is more likely to be limited by nitrate availability.

Wetland soils develop high levels of organic matter because of their slope position and hydrologic condition. Frequently inundated soils will have lower rates of litter decomposition because the flow of carbon from litter to microbial populations is reduced under anaerobic conditions (Groffman, 1994). The interactive nature of oxygen, nitrate, and carbon control of denitrification means that more denitrification generally occurs in intermittently flooded sites than in permanently flooded conditions (Reddy and Patrick, 1984).

Denitrification, measured directly using the acetylene inhibition technique (Tiedje et al., 1989), accounts for substantial nitrate loss from some riparian ecosystems. Denitrification has been identified as the key nitrate removal mechanism in several riparian forest buffer studies (Jacobs and Gilliam, 1985b; Pinay and Decamps, 1988; Correll and Weller, 1989; Groffman et al., 1992; Haycock and Pinay, 1993;

Jordan et al., 1993). Estimates in the range of 30 to 40 kg N ha^{-1} yr^{-1} have been reported for natural riparian forests in the U.S. (Hendrickson, 1981; Hanson et al., 1994a, Lowrance et al., 1984b). In several studies of denitrification in riparian ecosystems, denitrification has been concentrated in surface soil and rates are generally much lower below the top 12 to 15 cm of soil (Hendrickson, 1981; Groffman et al., 1992; Ambus and Lowrance, 1991; Lowrance, 1992). Schipper et al. (1993) reported very high denitrification in the top 30 cm of an organic riparian zone soil in New Zealand. Denitrification rates (measured on soil slurries made anaerobic with Argon gas) were over 11 kg N ha^{-1} d^{-1} in this site. This is likely an overestimate of actual denitrification because the slurries were made anaerobic. The denitrification rates measured were 1-3 orders of magnitude greater than most estimates in the literature. Measurements of denitrification in these organic soil zones showed that the denitrification was greatest at the upslope edge of the riparian zone where nitrate-enriched water entered the organic riparian soil (Cooper, 1990). These studies indicated that most of the organic riparian soils in the watershed were denitrifying at rates below their maximum capacity and could denitrify more if nitrate loadings increased (Cooper, 1990; Schipper et al., 1993). Denitrification is likely to be most important in wetland soils such as would be found in Zone 1 and some Zone 2 areas in the Chesapeake Bay watersheds (Lowrance et al. 1984b, Peterjohn and Correll 1984, Jacobs and Gilliam 1985b, Ambus and Lowrance 1991) but can also be significant in drier forest soils subject to high nitrate loadings and in grass vegetated filter strips (GVFS) (Ambus and Lowrance, 1991; Groffman et al., 1991).

While the factors regulating denitrification in surface soils and aquifers are relatively well understood, the amounts of direct denitrification of groundwater-borne nitrate are much less well established. Subsurface denitrification has been observed in several studies (Truedell et al., 1986; Slater and Capone, 1987; Smith and Duff, 1988; Francis et al., 1989; Obenhuber and Lowrance, 1991), yet other studies have found the potential for denitrification in the subsurface to be low or non-existent (Parkin and Meisinger, 1989; Ambus and Lowrance, 1991; Groffman et al., 1992; Bradley et al., 1992; Lowrance, 1992; Yeomans et al., 1992; Starr and Gilham, 1993). Subsurface microbial activity is usually limited by carbon availability. In settings where the total and dissolved carbon contents of aquifers are low, they are poor quality substrates for microbial growth (Lind and Eiland, 1989; Hiscock et al., 1991; Johnson and Wood, 1992; McCarty and Bremner, 1992) and anaerobic conditions necessary for denitrification to proceed are not generated.

Microbial attenuation of organic compounds arises from their ability to degrade these compounds as food sources or through non-energy yielding "cometabolism" reactions. There are many different microbial degradation mechanisms including aerobic, anaerobic, chemoautotrophic and heterotrophic pathways. The wide range of environments and high diversity of microbial metabolism in RFBS, should support many of these mechanisms. Further research into specific management strategies to foster a wide range of degradation strategies is needed (Paterson and Schnoor, 1992).

In many cases, riparian zone retention of groundwater-borne pollutants may depend on a complex interaction of hydrology, plant, soil and microbial factors. The potential importance of these interactions is hypothesized based on studies where significant rates of nitrate removal from groundwater were measured, but the potential for denitrification in the subsurface was low. Groffman et al. (1992) and Hanson et al. (1994a,b) suggested that surface soil denitrification of groundwater derived nitrate is an important route of N removal in riparian forests. This route depends on plant uptake of nitrate from groundwater, decomposition and N release from plant litter, and nitrification and denitrification of this N in surface soil. In riparian forests where this route of N removal is important, the nitrate removal function may depend on complex interactions between hydrology, plant dynamics, and soil microbial processes. These interactions vary within and between riparian forests and should be strongly influenced by soil drainage class, vegetation and soil type, climate, and groundwater quality. Although soil denitrification should be sustainable indefinitely under proper conditions with a supply of nitrate and available C, Hanson et al. (1994b) found that long term groundwater nitrate loading led to symptoms of N saturation in the surface soils of a riparian forest buffer.

d. Removal of Surface-borne Pollutants

Fewer studies have been published on NPS pollutant removal from surface runoff in Zone 2 type forests. The primary function of Zone 2 relative to surface runoff is to remove sediment and sediment-borne pollutants and to infiltrate runoff. Daniels and

Gilliam (in press) found that mature riparian forests, analogous to Zone 2 vegetation, were effective for sediment load reduction with removal of 50 to 80% of inputs from upland fields. Sediment trapping in riparian forest buffer zones is facilitated by physical interception of surface runoff that causes flow to slow and sediment particles to be deposited. Effective sediment trapping requires that runoff be primarily sheet flow. Channelized flow is not conducive to sediment deposition and can actually cause erosion of the RFBS. Two studies on long-term sediment deposition in riparian forests (Cooper et al., 1987, Lowrance et al., 1986, Lowrance et al., 1988) indicated that long-term deposition is substantial. In both these studies, two main actions occur: 1) the forest edge fostered large amounts of coarse sediment deposition within a few meters of the field/forest boundary; 2) finer sediments are deposited further into the forest and near the stream. Both Cooper et al. (1987) and Lowrance et al. (1986) found much higher depths of sediment deposition at the forest edge than near the stream. A second peak of sediment depth was often found in Zone 1, possibly from upstream sediment sources deposited in overbank flows (Lowrance et al. 1986). The surface runoff which passes through the forest edge environment is much reduced in sediment load because of coarse sediment deposition but the fine sediment fraction is enriched relative to total sediment load. These fine sediments carry higher concentrations of labile nutrients and adsorbed pollutants (Peterjohn and Correll, 1984; Magette et al., 1989) which are carried further into the riparian forest and are deposited broadly across Zone 2.

Movement of nutrients through Zone 2 in surface runoff will be controlled by a combination of: 1) sediment deposition and erosion processes; 2) infiltration of runoff; 3) dilution by incoming rainfall/throughfall; and 4) adsorption/desorption reactions with forest floor soil and litter. Studies that separate the effects of these various processes are not available. Peterjohn and Correll (1984) found large reductions in concentrations of sediment, ammonium-N, and ortho-P in surface runoff which passed through about 50 m of a mature riparian forest in the Maryland Coastal Plain, analogous to Zone 2. Although the concentrations of these pollutants were reduced by a factor of 3 or 4 in most cases, the flow-length was about twice that recommended in the RFBS specification. Daniels and Gilliam (in press) found that dissolved ortho-P loads in surface runoff were not reduced markedly in a Zone 2-like area of riparian forest. The studies of surface runoff through riparian forests agreed on the importance of eliminating channelized flow through the riparian forest and recommended spreading flow before it reached the forest buffer. Flow spreading is recognized as primarily a Zone 3 function in the RFBS specification. In-field practices are also critical in preventing channelized flow from reaching the field edge.

3. Zone 3—Sediment Removal and Spreading of Surface Runoff

The primary functions of Zone 3 are to remove sediment and sediment associated chemicals and to spread surface runoff entering as concentrated flow. Functions of grass vegetated filter strips (GVFS), analogous to Zone 3 of the RFBS, have been evaluated in a number of replicated experiments. Most of the available research on GVFS is applicable to evaluating the potential for sediment deposition in Zone 3 of the RFBS.

Several short-term experimental studies have found that GVFS were effective for removal of sediment and sediment-bound pollutants with trapping efficiencies exceeding 50% if flow was shallow (< 5 cm depth) (Young et al. 1980, Magette et al. 1987, Dillaha et al. 1989a). Magette et al. (1989) and Dillaha et al. (1989a) evaluated relatively narrow filter strips (4.6 m and 9.2 m) for control of nutrients and sediment moving from row-crop plots. Magette et al. concluded that: 1) the performances of GVFS were highly variable; 2) GVFS were more effective in removing suspended solids than in removing nutrients; 3) GVFS become less effective as more runoff events occur; and 4) the effectiveness of GVFS decreased as the ratio of GVFS length to source area decreased. Dillaha et al. (1989a) reached similar conclusions. They found that GVFS were effective immediately after establishment, removing up to 98% of the incoming sediment and that removal of incoming sediment bound total N and total P was nearly as effective. Soluble N (both NO_3 and NH_4) and soluble P were not removed effectively. Both Magette et al. (1989) and Dillaha et al. (1989a) conclude that narrow GVFS would probably have relatively short useful life spans. Dillaha et al. (1989a) reported that one GVFS was nearly inundated with sediment during the span of 6 rainfall simulator events. The sediment trapping efficiency fell from 90% in run 1 to 5% in run 6. GVFS were much less effective when flow was concentrated than when surface runoff was in shallow sheet flow (Dillaha et al.,1989a). Properly managed

Zone 3 areas are likely to perform similarly to GVFS in these experimental studies. Management of these areas will likely include periodic removal of sediment, reestablishment of vegetation, and removal of ephemeral channels.

Trapping efficiencies for sediment decrease at high runoff rates because of increased depth of flow (Barfield et al., 1979; Schwer and Clausen, 1989). Concentrations of total N, total P, suspended solids, and BOD were reduced up to 80% in feedlot runoff passed through GVFS ranging from 92 to 262 m (Dickey and Vanderholm, 1981). The need for relatively long filter strips was confirmed in other studies looking at runoff from chicken manure application areas (Bingham et al., 1980; Overcash et al, 1981). They found that the ratio of buffer area to land application area in order to achieve complete removal of contaminants in water leaving the GVFS was about 1:1. Therefore a filter area would need to be as large as the source area. This situation is often not possible due to inadequate land for filter areas or competition between land for GVFS and land for crop production. A buffer source area/length ratio of less than 1:1 would be adequate for less than complete removal.

Trapping efficiencies for sediment and nutrients also decrease when runoff enters the GVFS in concentrated flow (Dillaha et al., 1986). When this is the case, very little of the filtration capacity of either the GVFS or the riparian forest is used. If field practices do not eliminate channelized flow, it should be eliminated as near the upslope border of the RFBS as possible. The RFBS specification suggests using level-lipped spreaders to convert concentrated flow to sheet flow before it reaches Zone 2 (Welsh, 1991). These spreaders, when needed, would be part of Zone 3 so they could be managed (cleaned out and periodically restored) using farm equipment. Franklin et al. (1992) reported on the use of a level spreader to spread flow from agricultural fields before it entered a downslope forest filter zone (FFZ). Although they did not compare the FFZ with and without spreaders for natural rainfall events, they did compare hydrologic response with and without spreaders for simulated runoff events. Without a spreader, the time to reach peak flow at a flume below the FFZ was about 10 minutes and the time to reach zero flow below the FFZ after the water supply was cutoff was only 20 min. In contrast, with the level spreader in place, this artificial runoff took 45 minutes to reach a peak flow and 135 minutes to stop flowing after water was cut off (Franklin et al., 1992). Although specific water quality data are not available from this study with and without spreaders, spreading the flow affected the timing of the event with a smaller effect on peak and total flows.

Used as part of the RFBS, GVFS should substantially reduce sediment and sediment-borne pollutant loads reaching the stream. Improperly installed GVFS may serve to accentuate channelization problems in the landscape, leading to erosion of the forested zones of the buffer. For example, in an analysis of existing grass GVFS on 33 farms in Virginia (Dillaha et al. 1986, 1989b) found that sediment trapping was quite poor in many cases. In hilly areas, sediment trapping was generally low because runoff usually crossed the GVFS as concentrated flow. Rapid (1-3 years) accumulation of sediment caused several GVFS to become vigorous sediment producers. In cases where sediment accumulation was significant, runoff flowed parallel to the GVFS until a low point was reached where it crossed the GVFS as concentrated flow. Due to the uncertainties in long-term performance of GVFS, overall buffer efficiency and sustainability should be significantly increased by using a combination of grass strip and forest buffer as described here.

4. Integrated Water Quality Functions of the Three Zone Buffer System

Although no published studies of an integrated three zone buffer system are available, the studies cited above provide useful insights into the potential functions of each zone. Even with an integrated three zone system, it is possible that there will be conflicting objectives relative to the types of water quality functions desired from the RFBS.

Perhaps the most basic potential conflict relative to NPS pollution control with RFBS is the need to simultaneously control at least three major transport mechanisms of waterborne pollutants. It is likely that control of pollutants transported in the sediment adsorbed phase of surface runoff, the dissolved phase of surface runoff, and groundwater (dissolved phase only) may be optimal on different sorts of RFBS with differing soils, vegetation, and management. Riparian forest buffers must be effective in controlling multiple nonpoint sources of pollution.

Phillips (1989) proposed a general model of riparian buffer effectiveness based on detention time of surface and subsurface runoff and comparison to a reference buffer of known detention time and known effectiveness. Comparisons to a reference buffer with an assumed first order rate constant for nitrate removal were done. The detention-based model indicated that relatively flat, sandy, well-drained soils with high infiltration capacities would be the most effective buffers for nitrate removal. This approach is lacking relative to nitrate retention because it disregards the effects of different soils on denitrification and the unequal partitioning of nitrate between surface runoff and subsurface transport paths. The detention model (Phillips, 1989) correctly concludes that for surface-borne pollutants, increasing infiltration in the RFBS will be an effective measure for both dissolved and adsorbed pollutant control. Conversely, the sandy well-drained soils which have highest infiltration will likely have lowest denitrification rates and may have rapid groundwater movement rates leading to high rates of nitrate transport through the riparian forest buffer. This type of situation is described by Correll et al. (1994) for the entire riparian buffer and by Cooper (1990) for the mineral soils in the riparian zone.

For nitrate removal via denitrification, a riparian ecosystem where high nitrate water moves into high organic matter soils or subsoils is the best way to promote denitrification and the best way to permanently remove nitrate from the soil-water-plant system. This is illustrated both by the New Zealand riparian studies of organic riparian soils (Cooper, 1990; Schipper et al., 1993) and by the findings that denitrification is highly stratified in mineral soils with most denitrification occurring in the high organic carbon surface soils (Ambus and Lowrance, 1992; Hanson et al, 1994a). Organic rich wetland soils can often respond to increased nitrate loads with increased denitrification (Groffman, 1994). The same conditions which are likely to promote denitrification are likely to decrease the amount of retention of surface-borne pollutants. Wetland soils which are frequently inundated will have little or no infiltration capacity or available water storage capacity.

II Riparian Forest Buffer Systems in Physiographic Provinces of the Chesapeake Bay Watershed

A. COASTAL PLAIN

1. General Land Use and Hydrology

The Coastal Plain has higher proportions of both cropland (32%) and wetland (21%) than any other physiographic province of the Bay Watershed (Table 1). The Coastal Plain portions of the CBW are comprised of watersheds with low topographic relief, relatively high moisture infiltration capacities, well-distributed rainfall throughout the year, and unconfined surficial aquifers. Streamflow is mainly derived from groundwater discharge from the surficial aquifer. Direct surface runoff in agricultural watersheds generally accounts for about 5 to 15% of streamflow (Peterjohn and Correll, 1984; Staver et al., 1988). The remainder of the precipitation either infiltrates and is available for either groundwater recharge or evapotranspiration or goes directly into surface water as stream or detention storage. Although this general view of the Coastal Plain is useful, variations in soils, topography, subsurface stratigraphy, and land use within the Coastal Plain control the fate of NPS pollutants relative to RFBS.

The CBW Coastal Plain is often divided into Inner and Outer Coastal Plains. The Inner Coastal Plain is mostly the western shore of Chesapeake Bay and the upper Eastern Shore. The Outer Coastal Plain is primarily the lower Eastern Shore/Delmarva Peninsula. Inner Coastal Plain areas have relatively high topographic relief compared to Outer Coastal Plain systems and generally have finer textured, nutrient-rich soils compared to the nutrient-deficient, sandy soils of the Outer Coastal Plain (Correll et al., 1992). A more detailed classification of the Coastal Plain was developed by the U.S. Geological Survey for the Delmarva Peninsula (Phillips et al, 1993). This classification of hydrogeomorphic regions was based on qualitative analysis of geologic and geomorphic features, soils, drainage patterns, and land cover (Figure 3). The upland, non-tidal area of the Delmarva was divided into Inner Coastal Plain which closely correlates with the Inner Coastal Plain of Correll et al. (1992) and three Outer Coastal Plain hydrogeomorphic regions: well-drained upland, poorly drained upland, and surficial confined region. Differences in the physical characteristics of these regions result in variations in the functions of RFBS within them. The following discussion presents the general hydrogeomorphic characteristics associated with each region.

a. Inner Coastal Plain

The Inner Coastal Plain (ICP) includes the portion of the Coastal Plain located on the western shore of Chesapeake Bay and the area immediately south of the Fall Line on the Delmarva Peninsula. Tidal sections of rivers extend far into the ICP, near the Fall Line in some cases. Watersheds in the ICP are characterized by well-drained soils on uplands with poorly drained soils limited to riparian zones. Land use is primarily agricultural on uplands and forested in riparian zones. Topography of this region is gently rolling with a high degree of stream incision.

The ICP is a hydrologically complex region because sands and gravels that comprise the surficial aquifer are thin and overlie subcropping sands or finer-textured confining beds of older Coastal Plain aquifers. Stream valleys are commonly incised into the older units. As a result of this configuration, the surficial deposits do not form an extensive aquifer as they do in other parts of the Coastal Plain. Shallow groundwater flow systems in the surficial sediments commonly extend from topographic highs into the deeper aquifer where they are close to the surface. In addition, older water from deeper aquifers often discharges upward to streams. If the surficial aquifer overlies a shallow confining bed, groundwater flow is restricted to shallow depths where it comes into contact with riparian zone sediments and soils near aquifer discharge areas.

The Rhode R. Watershed along the western shore of Maryland is representative of the hydrologic conditions common to much of the ICP. This 2286 ha watershed is 62% forest, 23% croplands, 12% pasture, and 3% freshwater swamp (Jordan et al., 1986). The watershed is underlain by a relatively impermeable clay layer which forms an effective aquiclude. Most groundwater flow to streams is in a shallow surficial aquifer (Correll 1983). The 160 yr average rainfall is 108 cm. The long-term average precipitation by season is 28 cm, 31.4 cm, 24.5 cm, and 24.6 cm for December to February, March to May, June to August, and September to November, respectively (Higman and Correll, 1982 cited in Peterjohn and Correll, 1984). For the Rhode R. Watershed, slow streamflow (baseflow or groundwater discharge) averaged 29.6 cm of flow while quickflow (mostly stormflow or surface runoff from all contributing areas) accounted for 4.97 cm (Correll, unpublished in Peterjohn and Correll, 1984). Studies on Rhode R. indicated that 86% of all watershed discharge comes from slow flow or groundwater discharge and 14% from direct surface runoff. For one year of study March, 1981 to March, 1982, Peterjohn and Correll (1984) estimated that about half of all quickflow took place in the Summer (June to August) and that over half of slow flow (groundwater discharge) took place in winter.

b. Well-Drained Upland

Watersheds in the well drained upland (WDU, Figure 3) are characterized by predominantly well-drained soils on uplands and poorly drained soils on floodplains in stream valleys. The topography is relatively flat to gently rolling and there is a high degree of stream incision (Phillips et al., 1993). Most of the upland area is used for agricultural crop production with wooded areas generally confined to narrow riparian zones. Sediments of the surficial aquifers are primarily sand and gravel and range from about 6 to 12 m in the north to 24 to over 30 m thick in the south (Owens and Denny, 1979). The aquifer is unconfined and the depth to water ranges from 3 to 10 m beneath topographic highs, to land surface in surface-water discharge areas.

Groundwater flow paths range from about 1 km to several km in length in the well-drained upland (Shedlock et al., 1993). The longest; oldest flow paths originate at topographic highs, extend to the base of the aquifer, and discharge to 2nd and 3rd order streams through the hyporheic zone (beneath the stream channel). The water contained in them is generally less than 50 years old near aquifer discharge areas (Dunkle et al., 1993). Shorter, younger flow paths originate in near-stream recharge areas and are the main source of baseflow to first-order streams.

c. Poorly-Drained Upland

Watersheds in the poorly drained uplands (PDU, Figure 3) are characterized by interspersion of poorly drained areas with forested land use, and moderately well-drained and well-drained areas with agricultural use (Shedlock et al., 1993). In the northern part, the region has hummocky topography and low relief with many seasonally ponded wooded depressions. In the southern part, topography is relatively flat with broad poorly drained forested areas that are seasonally flooded (J. M. Denver, unpublished). Streams are small and sluggish in the poorly drained upland and flow through shallowly-incised valleys with low gradients (Phillips et al., 1993). Riparian zones are usually forested and often contain wetlands. Some parts of the poorly drained upland have been ditched to promote drainage of agricultural fields.

Sediments that make up the surficial aquifer in the PDU are predominantly sands and gravels, similar to those in the well-drained upland. The sediments range in thickness from about 8 m in the north to more than 30 m in the south (Owens and Denny, 1979). The water table is usually within 3 m of the land surface. This region is characterized as poorly drained because of the combination of regionally high water table and small degree of stream incision that results in groundwater gradients too low to effectively drain the region, rather than a low permeability substrate (Phillips et al., 1993).

Except for areas immediately adjacent to streams, groundwater flow paths in the PDU range from about 100 m to about 1 km in the northern part of the region where the aquifer is thin. In the southern part, where the aquifer is thick, flow paths are up to several km in length and generally originate near the regional drainage divide. Local flow patterns vary seasonally, however, smaller localized flow paths associated with the depressional wetlands and intermittent streams in the north and intermittent streams in the south are active during wet seasons (generally winter and spring). A more regional flow system from topographic highs to perennial streams is active throughout the PDU during the drier seasons (generally summer and fall).

d. Surficial Confined

Watersheds in the surficial confined (SC, Figure 3)

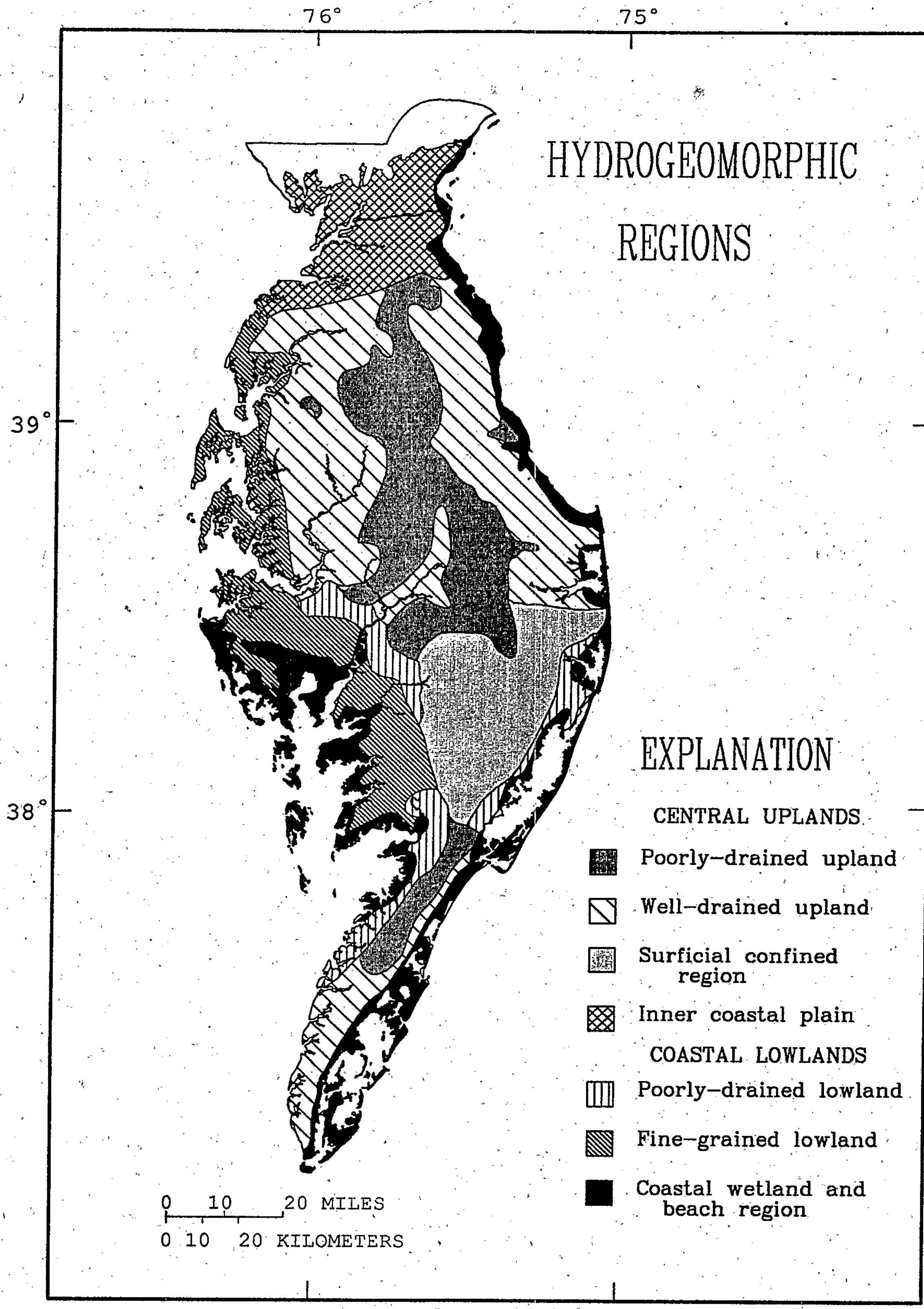

FIGURE 3. *Hydrogeomorphic regions of the Delmarva Peninsula. (From Phillips et al., 1993).*

region are geomorphologically similar to the southern part of the poorly drained upland with low relief and shallow incision of stream valleys, features that contribute to the poor drainage in the region. Topographically, the area is a flat sandy plain with low ridges that rise a few meters above the surrounding landscape. The plain is dominated by poorly drained soils and the ridges are dominated by well-drained soils. Throughout the region large tracts of forest are interspersed with agricultural fields on the plains; there are broad forested riparian zones and swamps around the major drainageways. With the exception of the sandy dune ridges, agricultural land is heavily ditched to promote soil drainage and would probably be forested wetlands in the absence of ditching (Phillips et al., 1993).

The surficial aquifer is geologically heterogeneous in the region, consisting of a major sand unit 25 to 30 m thick overlain by 0 to 13 m of complexly layered clay, slit, and peat, which is itself overlain by 1 to 6 m of wind-deposited sand with some peaty sand, slit, and clay lenses at the base (Owens and Denny, 1979). The complex of fine-grained deposits acts as a confining unit between the sands of the upper and lower units, except some areas where it is absent or entirely composed of sand. The water table is generally less than 3 m below land surface and occurs in the upper sand unit. Local groundwater flow paths, in the upper unit, are relatively shallow and generally less than 300 m long and extend from water-table highs in interfluves between ditches and streams into the ditches and streams. Regional groundwater flow paths, in the lower units, are up to ten kilometers long and extend from the uplands near the regional drainage divide to major streams and rivers. Local and regional flow paths are separated in most areas by the confining layer, but local heads are higher than regional heads in most places, and shallow flow paths extend into the lower sand where confining beds are absent (Shedlock et al., 1993). Residence time in the upper sand is 15 years or less; in the deeper unit, groundwater residence time is at least 40 to 50 years, except where there is hydraulic connection with the shallow unit (Dunkle et al., 1993).

2. Control of Nonpoint Source Pollutants

Although more studies have been done on Coastal Plain riparian forests than in other physiographic regions, a number of questions remain about the NPS pollution control capacity of naturally occurring riparian forest buffers. Other questions remain about the NPS pollution control capacity of reestablished and managed RFBS. The following discussion will necessarily focus primarily on what is know about naturally occurring riparian forest buffers and experimental GVFS. Although discussion of reestablished RFBS will be limited, a number of useful conclusions can be drawn from the existing Coastal Plain information.

The studies on riparian forest buffer effects on NPS pollutants in the Coastal Plain have tended to concentrate on the fate of nitrate in groundwater, with a secondary emphasis on the fates of N, P, and sediment in surface runoff. Three areas of the Coastal Plain (Georgia, Maryland, and North Carolina) have been studied where gaged watersheds were used as the basis for nutrient budget estimates of riparian forest buffers. The studies from Maryland (Rhode R.) have been used to develop nutrient budgets for watersheds and riparian systems (Peterjohn and Correll, 1984; Jordan et al., 1986; Correll and Weller, 1989; Correll et al., 1992). The studies from Georgia (Little R.) have been used to develop both nutrient and sediment budgets (Lowrance et al., 1983, 1984a,b, 1985; Fail et al., 1986). The studies from North Carolina have been used to develop nitrate budgets for riparian systems (Jacobs and Gilliam, 1985a,b). Hydrologic conditions for all of these studies were representative of ICP conditions.

A second general type of study has been conducted on the fate and/or transport of potential NPS pollutants, primarily plant nutrients and sediment. These studies have also been primarily in Maryland and Delaware (Correll et al., 1993; Jordan et al., 1993; Whigham et al., 1986), Georgia (Lowrance et al., 1988; Ambus and Lowrance, 1991; Lowrance, 1992; Vellidis et al., 1993), and North Carolina (Cooper et al., 1987; Cooper and Gilliam, 1987). In addition, there are several studies of Coastal Plain hydrology or water quality which provide information on upland riparian interactions or provide limited data on NPS removal in riparian forest buffers. These are studies which, in general, were not designed specifically to examine the removal of potential NPS pollutants in riparian forest buffers (Lowrance and Leonard, 1988; Weil et al., 1990; Staver and Brinsfield, 1990). Preliminary results on integrated grass and forest buffers in the Coastal Plain have been published (Parsons et al., 1991, 1994) and detailed studies of GVFS have been conducted in the Coastal Plain of Maryland (Magette et al., 1989).

a. Nutrient Budgets for Riparian Forests

The most direct means of determining the NPS pollution control function of a riparian forest is to develop annual or longer term mass balances. Developing nutrient or sediment budgets requires a watershed from which hydrologic measurements can be made which assure that all watershed outputs are measured and sampled. If the riparian forest buffer is continuous around the entire stream system and groundwater discharging to streams moves through riparian soils and shallow sediments, the streamflow output can be treated as the output from the riparian forest system. The inputs to the riparian system must be estimated from sampling of groundwater and surface water inputs. The studies which have done this for Coastal Plain riparian forests are summarized in Table 2. Total N and total P retention have been estimated in studies of Watershed 109 (WS-109) of the Rhode R. in Maryland and the Heard Creek tributary of Little R. in Georgia. Both of these Coastal Plain systems have effective aquicludes at depths which limit recharge to deep groundwater and which cause all or nearly all excess precipitation to move through riparian systems and exit the watershed as streamflow.

Estimates of N retention were 89% of input (Rhode R.), and 66% of input (Little R.). P retention in Rhode R. was slightly less (80% of input) but much less in Little R. (24% of input). Total N and P budgets for Little R. (Table 2) did not include surface runoff inputs of N and P from the agricultural areas to the riparian forest but did include all streamflow outputs of N and P. Streamflow includes surface runoff which moved through the riparian forest and contributed to stormflow. Therefore, the N and P retention (input-output) estimates for the Little R. site are underestimates of the actual retention. Peterjohn and Correll

TABLE 2.
Total nitrogen, nitrate-nitrogen, and total phosphorus budgets for riparian forest ecosystems in the Coastal Plain

Reference	Location	Input	Output	Retention+	Flux Notes*
		kg ha^{-1} yr^{-1}			
Total N					
Peterjohn Correll, 1984	Rhode R., MD	83	9	74	NO_3, NH_4, Org-N in & SRO, GW, P, PSF, PQF.
Lowrance et al., 1983	Little R., GA	39	13	26	NO_3, NH_4, Org-N in GW, P, SF.
Nitrate-N					
Correll & Weller, 1989	Rhode R., MD	45	6.4	38.6	NO_3, in GW, SF (baseflow only).
Lowrance et al., 1983	Little R., GA	22	2.1	19.9	NO_3 in GW, SF.
Cooper et al., 1986	Beaverdam Cr., NC	35	5.1	29.9	NO_3 in GW, SRO, SF.
Total-P					
Peterjohn & Correll, 1984	Rhode R., MD	3.6	0.7	2.9	Total P in SRO, GW, P, PSF, PQF.
Lowrance et al., 1983	Little R., GA	5.1	3.9	1.2	Total P in GW, P, SF.

+Retention = Input-Output

*SRO = surface runoff input; GW = groundwater input; P = precipitation input; SF = streamflow output; PSF = partitioned slowflow; PQF = partitioned quickflow

(1984) included direct estimates of both surface runoff and groundwater inputs and outputs for Rhode R. Their budget estimates were based on these direct measurements rather than streamflow outputs. Streamflow outputs for Rhode R. were different than the riparian budget output for both total N and P. This difference has only a negligible effect on the total N budget, but has a large effect on the total P budget. If streamflow outputs are considered the output from the riparian forest for the Rhode R. site, the total N retention is still 83% of inputs, but P retention is zero.

The Little R. and Rhode R. studies were both done in systems which are likely to maximize retention by natural riparian forests. Although the studies report different ranges of percent retention for N and P, retention of N was generally high. Both watersheds have percentages of agricultural land typical for the more agricultural portions of the Coastal Plain and are representative of potential inputs to riparian systems in the absence of animal confinement facilities and manure disposal systems. These natural riparian systems would appear to retain at least two-thirds of the N inputs but perhaps as little as one-third of the P input.

In both the Rhode R . and Little R. studies, nitrate in subsurface flow made up the majority of total inputs to the riparian forest system. The input in groundwater for WS-109 of Rhode R. in the year reported on in Peterjohn and Correll (1984) was 57 kg NO_3-N ha^{-1} yr^{-1} based on the area of riparian forest. This accounted for 69% of the total N input (Table 2). Based on two more years of data for WS-109 of Rhode R., the input averaged 45 kg NO_3-N ha^{-1} yr^{-1} (Correll and Weller, 1989). Data from Little R. showed that groundwater input was 22 kg NO_3-N ha^{-1} yr^{-1}, 56% of total N input. A third study of nitrate budgets (Cooper et al., 1985) on a Coastal Plain watershed in North Carolina showed similar results to the MD and GA studies. Nitrate retention rates of 85%, 86%, and 90% for the three studies (NC, MD, GA, respectively) reflect removal of nitrate through both denitrification and plant uptake. Plant uptake (and perhaps microbial immobilization) contribute to transformation of a predominately nitrate input to the riparian zone into a predominately organic N output in streamflow. Total N input to the riparian forest on Rhode R. was 69% nitrate. Streamflow was 51% organic N (Correll et al, 1992, Correll, 1983). On the Little R., groundwater inputs to the riparian forest were 74% nitrate. Streamflow outputs were 18% nitrate and 80% organic N. A later study of the entire Little R. watershed showed consistent trends of nitrate increase during stormflow, indicating that the nitrate removal/transformation capacity of riparian forests is partially by-passed when water moves through more quickly during high flows (Lowrance and Leonard, 1988).

b. Removal of Nitrate from Groundwater

Although elemental, nutrient, chemical, and sediment budgets on a watershed scale are the most complete way to evaluate the functions of riparian forest buffers and offer the best information on potential load reductions, a number of studies have examined nitrate concentration changes in riparian forests. This emphasis on nitrate is due to a number of factors including the relatively high transport rate of nitrate from most agricultural systems, the availability of nitrate for algal uptake as a stimulus for eutrophication, and possible impacts on downstream or shallow groundwater drinking water supplies. Studies in at least five separate Coastal Plain locations have examined the changes in nitrate concentrations as shallow groundwater moves from agricultural fields through naturally occurring riparian forests (Figure 4). Studies in four separate locations have shown that average annual edge-of-field nitrate levels of 7 to 14 mg NO_3-N L^{-1} decreased to 1 mg L^{-1} or less in shallow groundwater near streams. Some studies have used chloride concentrations and nitrate:chloride ratios to separate the effects of dilution from the effects of biological removal of nitrate. Decreases in chloride concentrations were generally small compared to nitrate decreases. Chloride concentrations usually increased at some point in the shallow groundwater system probably due to exclusion of Chloride from the transpiration stream (Peterjohn and Correll, 1986; Jordan et al., 1993; Correll et al., 1993; Lowrance, 1992).

Most studies of nitrate dynamics in riparian forests have shown that removal of nitrate from groundwater continued year-round. Mechanisms to explain this have not been elucidated, although it is likely that in some of the Southeastern Coastal Plain areas, relatively warm soils and evergreen or tardily deciduous (broad-leaf trees that lose leaves in the spring) vegetation can provide biological removal of the nitrate. Most Coastal Plain areas of the CBW have lower groundwater and soil temperatures in the winter and little or no evergreen vegetation. Weil et al. (1990) observed year-round reductions of groundwater nitrate in streamside forests on tributaries of the Choptank R. on the Eastern Shore of Maryland. Groundwater

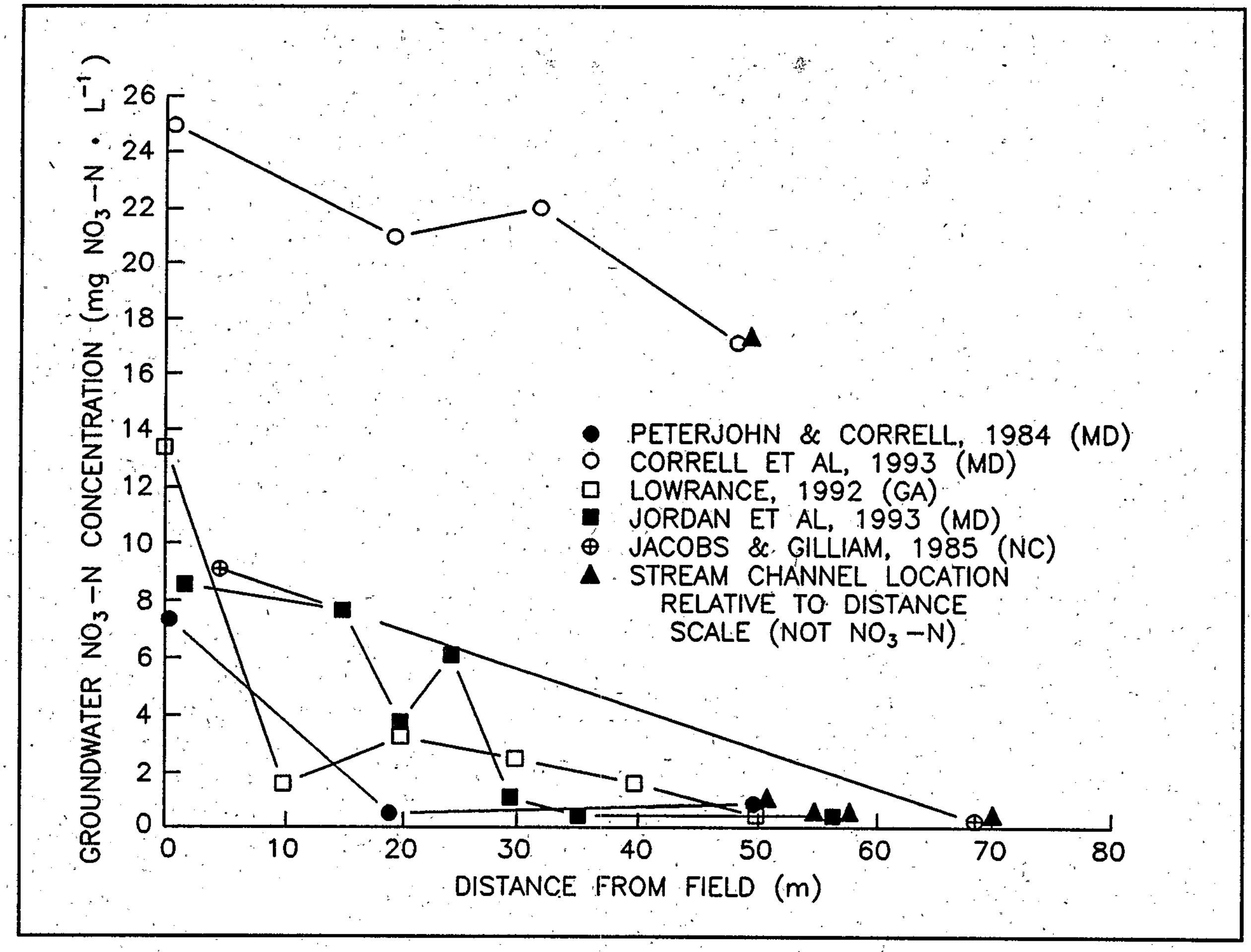

FIGURE 4. *Nitrate concentrations in groundwater beneath riparian forests from five Coastal Plain sites.*

under riparian forests always had less than 1 mg NO_3-N L^{-1} while adjacent fields had concentrations of 15-40 mg NO_3-N L^{-1}. The decreases in chloride concentrations were much less than the nitrate decreases. Year-round nitrate removal has been observed, but not explained.

At least one study has shown that in situations with relatively high nitrate concentrations entering from an adjacent field, substantial nitrate concentration reductions can occur but still leave high concentrations in shallow groundwater at the stream (Correll et al, 1993), (Figure 4). This site, on a tributary of the Choptank R. on the Delmarva Peninsula is located in the Well Drained Uplands. Nitrate concentration reductions were actually higher at this site than at two other Maryland Coastal Plain sites (Peterjohn and Correll, 1984; Jordan et al, 1993) but groundwater concentrations near the stream were 12 to 18 mg NO_3-N L^{-1}. Similar results were inferred from a study of nitrate in regional groundwater and nitrate levels in streamflow for the WDU hydrogeomorphic region (Phillips et al., 1993). In related work, Böhlke and Denver (in press) concluded that the riparian forest wetland next to the stream in the Locust Grove Watershed in Maryland had little effect on nitrate movement to the stream. Hydrologic data and groundwater flow modeling show that groundwater discharges upward directly to the streambed from the aquifer system, effectively bypassing the riparian zone (Reilly et al., 1994). Baseflow concentrations of nitrate commonly exceeded 9 mg NO_3-N L^{-1} in the stream draining this watershed, and isotopic analysis indicated that denitrification was not significantly affecting nitrate concentrations (Bohlke and Denver, in review).

Nitrate transport into tidal streams is often dominated by direct recharge through sediments in intertidal zones (Reay et al., 1992; Simmons et al., 1992;

Staver and Brinsfield, 1994). Approximately 80 kg ha^{-1} yr^{-1} of NO_3-N was discharged to a tidal creek in Maryland with apparently most groundwater moving at least 2 m below the ground surface in near-stream areas (Staver and Brinsfield, 1994). These situations may allow little chance for nitrate removal. The direct NO_3-N discharge to tidal streams make riparian buffers desirable (Simmons et al., 1992), if proper management could allow direct vegetation uptake from the groundwater.

c. Nutrient Removal Processes

Removal processes were quantified in most of the riparian forest research on nutrient budgets and nitrate transport. Studies in Maryland and Georgia have made direct estimates of N and P uptake by vegetation and storage of N and P in woody biomass. Estimates from Watershed 109 of Rhode R. (Peterjohn and Correll, 1984; Correll and Weller, 1989) indicated that total vegetation uptake of N and P was 77 and 10 kg ha^{-1} yr^{-1}, respectively. N and P storage in woody biomass was less than total uptake (Table 3).

Extensive data on total N and P uptake and woody storage were reported by Fail et al. (1986, 1987). Values for P uptake and storage are similar for the Little R. and Rhode R. studies (Table 3). Major differences between the two studies were found for N woody storage and N uptake. Fail et al. (1986, 1987) reported mean storage of N in wood as 52 kg N ha^{-1} yr^{-1}. The range was from about 35 to 98 kg N ha^{-1} yr^{-1}. The net primary productivity reported by Fail et al. and Peterjohn and Correll are similar as are leaf N concentrations and leaf and wood P concentrations. Wood N concentrations averaged 7900 $\mu g\ g^{-1}$ in the Little R. studies, compared to average sapwood values of about 900 $\mu g\ g^{-1}$ in the Rhode R. study. Fail et al. (1987) used branch wood samples to represent the entire woody biomass of the tree and so overestimated N accretion in wood. Based on a number of studies, they pointed out that bole wood N content averaged about 43% of branch wood N content. This correction would bring the net wood accumulation of N down to about 22 kg N ha^{-1} yr^{-1}. Total N uptake would be about 84 kg N ha^{-1} yr^{-1} if this correction is applied.

Denitrification has been shown to be an important N removal process in Coastal Plain riparian forests either: 1) through indirect measurement using the acetylene inhibition technique; 2) through measurement of environmental conditions which control denitrification (Eh, water-filled pore space, N and C availability) and verifying that proper environmental conditions exist; or 3) through measurement of denitrification potential (Ambus and Lowrance, 1991; Lowrance et al., 1984b; Hendrickson 1981, Jacobs

TABLE 3.
Above-ground woody vegetation uptake of N and P in Coastal Plain riparian forests.

		Nitrogen		Phosphorus	
Reference	Location	Total Input	Woody Storage	Total Uptake	Woody Storage
		kg ha^{-1} yr^{-1}			
Correll & Weller, 1989	Rhode R., MD	ND*	12 to 20	ND	3 to 5
Peterjohn & Correll, 1984	Rhode R., MD	77	12	10	1.7
Fail et al., 1986, 1987 (mean)	Little R., GA	114	52	7.5	3.8
Fail, 1986 (maximum)	Little R., GA	194.4	97.6	12.6	6.9
Fail, 1986 (minimum)	Little R., GA	80	34.6	4.5	1.9

* ND = not determined

and Gilliam, 1985b; Correll et al., 1994; Jordan et al, 1993, Lowrance, 1992). The general conclusion of all these studies was that denitrification occurred in most riparian forest soils, especially in the root zone, or that conditions were favorable for denitrification. Recent work by Böhlke and Denver (in press) indicated that denitrification can also occur in sediments beneath the influence of the riparian root zone.

Denitrification was measured in riparian forests of Little R., GA in conjunction with water quality and hydrologic measurements (Hendrickson 1981). A total of 1114 soil cores (0 to 10 cm) were taken monthly for a year from 6 riparian forest sites on the Heard Cr. tributary of Little R. Summarized data from these studies were used to estimate a denitrification rate of 31 kg N ha^{-1} yr^{-1} for the top 50 cm of soil for the entire riparian zone of the watershed (Lowrance et al., 1984b). Denitrification rates under conditions of high N and C subsidy from a swine operation ranged up to 295 kg N ha^{-1} yr^{-1} (Hendrickson, 1981). Lowest denitrification rates (1.4 kg N ha^{-1} yr^{-1}) were measured in a riparian zone adjacent to an old field which received no fertilizer application. Hendrickson (1981) found that the active cores (those producing N_2O above background levels) ranged from 11% to 66% of the cores taken, depending on the site. This study confirmed the potential for denitrification in surface soils as well as the high variability to be expected in field measurements of denitrification. Soil cores taken to 50 cm in 10 cm increments showed that, except near the stream channel, denitrification activity below 20 cm depth was much lower than activity in the top 20 cm.

Later studies from Little R., GA have also shown that denitrification potentials at the top of the water table are measurable, but very low (Lowrance, 1992). Nitrate which moves into upper soil layers is likely to be reduced by denitrification. Nitrate moving through a restored RFBS was reduced by high rates of denitrification averaging 68 kg N ha^{-1} yr^{-1}. These high rates were due to a relict forested wetland soil and movement of high nitrate water in the root zone (Lowrance et al., in press). In addition, nitrate which moves through anoxic sediments in riparian zones is also likely to be reduced. In contrast, nitrate in groundwater which moves through generally oxic aquifer material or nitrate which does not generally come in contact with the root zone soil layers is much less likely to be denitrified.

The interaction of vegetation nitrogen uptake, organic carbon production via litterfall and root senescence, and microbial denitrification appear to be driving nitrate removal in most Coastal Plain riparian forests. Correll and Weller (1989) proposed a model of belowground processes affecting nutrients (Figure 5) which conceptualized the system as being controlled largely by oxidation-reduction conditions. Organic matter from decomposing litter and roots

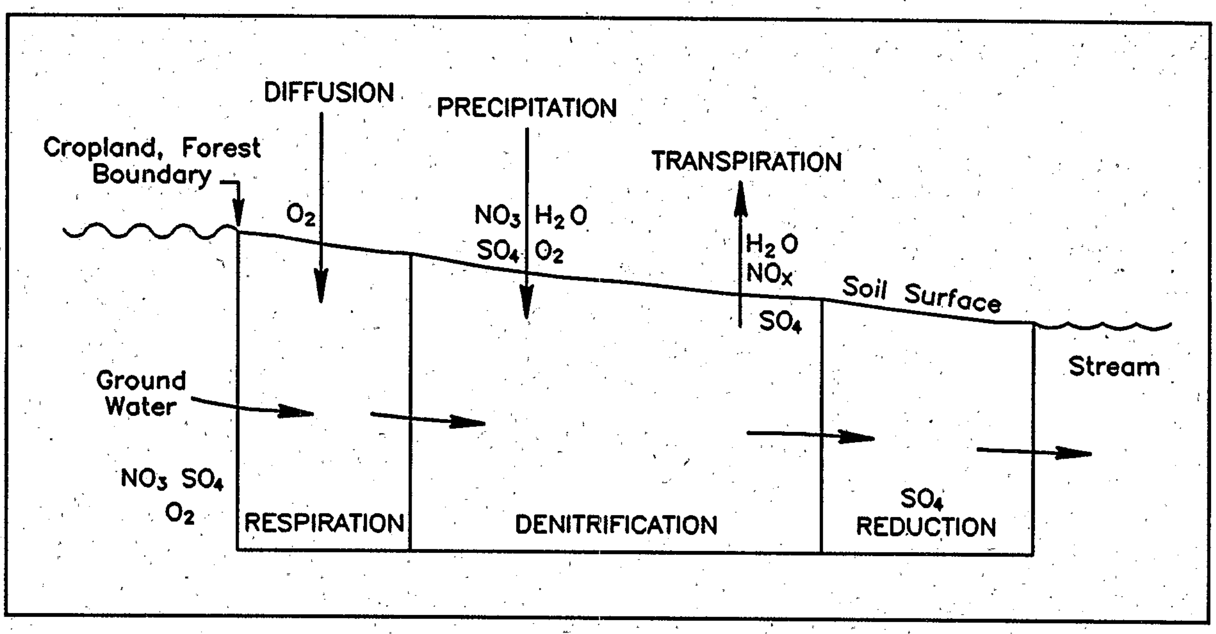

FIGURE 5. Conceptual model of below ground processes affecting groundwater nutrients in riparian forest (from Correll and Weller, 1989).

serves as an energy source and oxygen is consumed through aerobic respiration, followed by nitrate reduction, followed by sulfate reduction when conditions become sufficiently reduced. In the presence of carbon rich sediments or relict organic matter horizons, these processes could potentially proceed without forest vegetation. A similar conceptual model for nitrate removal in Coastal Plain riparian forests was proposed by Lowrance (1992). Stratified denitrification potential in riparian forests of Little R. indicated that denitrification coincided with the stratification of N and C from litter and roots. These findings support the hypothesis that nitrate removal in RFBS is dependent on interactions in the forest ecosystem rather than just a poorly drained soil adjacent to a stream. It is likely that nitrate removal in all Coastal Plain forest sites (where substantial removal has been demonstrated) was due to these complex interactions of vegetation and the belowground environment. It should be noted, however, that hydrologic conditions in which groundwater containing nitrate passes through or near the root zone must be present for this mechanism to operate effectively. Although most of the Coastal Plain studies of nitrate removal were in areas with relatively flat wetland soils near the stream, removal often took place in areas immediately downslope from the fields on better drained soils.

d. Removal of Sediments and Nutrients from Surface Runoff

Removal of nutrients and sediment from surface runoff in the RFBS will be a function of both Zone 3 and Zone 2. Sediment and nutrient deposition from surface runoff moving through a Coastal Plain riparian forest has been estimated from direct sampling of surface runoff in the Rhode R. watershed (Peterjohn and Correll, 1984). Estimates of sediment deposition have been made based on soil morphology and ^{137}Cs profiles in Little R., GA and in Cypress Creek, NC. GVFS have been widely studied, with at least one detailed study of fescue buffers in the Coastal Plain of Maryland (Magette et al., 1989).

The estimated range of sediment deposition rates in riparian forests is large and apparently somewhat dependent on estimation technique (Table 4). Although the different methods give widely divergent numbers, in all cases sediment deposition accounted

TABLE 4.
Sediment deposition in Coastal Plain riparian forests.

Reference	Location	Sediment Deposition	Notes
		Mg ha^{-1} yr^{-1}	
Peterjohn & Correll, 1984	Rhode R. (MD)	4.2	Annual measurements, first order stream, runoff samples
Cooper et al., 1987	Cypress Cr. (NC)	105-315*	^{137}Cs measurements—forest edge
Cooper et al., 1987	Cypress Cr. (NC)	35-105*	^{137}Cs measurements—ephemeral & Intermittent streams
Cooper et al., 1987	Cypress Cr. (NC)	0-35*	^{137}Cs measurements—floodplain swamp
Lowrance, et al., 1986	Little River (GA)	35-52	Watershed based, long term, sediment delivery ratio, soil morphology
Lowrance, et al., 1987	Little River (GA)	256-262	Single field/forest system ^{137}Cs measurements

*Based on sediment depths in Cooper et al. (1987) and assumed bulk density of 1.4 g cm^{-3}.

for 80 to 90% of gross erosion from the uplands. Relatively low overall deposition rates (4.2 Mg ha^{-1}) reported from direct sampling were associated with 90% reductions in sediment concentration in 19 m of flow through a riparian forest (Peterjohn and Correll, 1984; Table 6). Sediment deposition estimates need to be compared to the gross erosion rates from cropland with information on the contributing area:riparian area ratio. With a field:forest ratio of approximately 2:1, the riparian forest would attenuate cropland erosion rates of about 2.1 Mgha.$^{-1}$yr^{-1} per year (Peterjohn and Correll, 1984). This is well below the tolerance value for the upland soils and many fields would contribute higher sediment loads from erosion. In contrast, a sediment deposition rate of 35 Mg ha^{-1} yr^{-1} at a 2:1 field to forest ratio would attenuate erosion from cropland contributing up to 17 Mg ha^{-1} yr^{-1}. Very high sediment deposition rates (up to 315 Mg ha^{-1} yr^{-1}) reported from ^{137}Cs distribution studies (Table 4) were due to high deposition at field edge. This deposition was mostly coarse material and did not contain large amounts of adsorbed nutrients.

Sediment removal in GVFS in Coastal Plain areas is very effective in relatively short distances (Table 5). The RFBS would generally include a grass strip of a little more than 4.6 m. If concentrated flow occurs across the GVFS, sediment removal is much less efficient. The grass strips also become less effective when multiple rainfall events take place in a few days or when sediment begins to accumulate and forms berms which can lead to channelized flow (Magette et al., 1989). Field evaluations of GVFS indicated that they were more effective in Coastal Plain areas of Virginia than in steeper topography (Dillaha et al., 1989b). Slopes in Coastal Plain areas were more uniform and field reconnaissance indicated that significant portions of stormwater runoff entered the GVFS as shallow uniform flow. These GVFS needed regular maintenance (sediment removal and possible revegetation every 1 to 3 years) because of the amounts of sediment deposition (Dillaha et al, 1989b).

Nutrient removal from surface runoff has received very limited study (Tables 5 and 6). The 4.6 m filter strips used by Magette et al., in the Maryland Coastal Plain generally did not remove total N from surface runoff and removed only 27% of the total P load. The 9.2 m filter strips had total N and P removals of nearly 50%. Peterjohn and Correll (1984) reported concentration reductions of 74% for total N and 70% for total P in flow through 19 m of mature riparian forest in Watershed 109 of Rhode R. (Table 6). This width of forest would be very similar to a Zone 2 which conformed to the RFBS specification.

Data from Magette et al., (1989) and Peterjohn and Correll (1984) have been combined to estimate the effects of combined Zones 3 and 2 on sediment and nutrients in surface runoff (Table 6). The GVFS of Magette et al. are analogous to Zone 3 and the 19 m of mature forest from Peterjohn and Correll is analogous to Zone 2. These widths, 4.6 m and 19 m, are almost the exact widths specified in Welsch (1991) for Zones 3 and 2, respectively. Applying the 89.8% sed-

TABLE 5.
Inputs, outputs, and % removals of sediment (total suspended solids), total N (particulate + dissolved), and total P (particulate + dissolved) from experimental Ky 31-Fescue vegetated filter strips in Maryland Coastal Plain. From Magette et al., 1989.

Filter Strip Width	Total Suspended Solids			Total Nitrogen			Total Phosphorus		
	Input	Output	Removal*	Input	Output	Removal	Input	Output	Removal
m	---- Mg ha^{-1} ----		%	----kg ha^{-1}----		%	----kg ha^{-1}----		%
4.6	27.2	9.3	66	39.4	41.6	-5	32.3	23.6	27
9.2	27.2	4.9	82	39.4	20.7	47	32.3	17.4	46

*Removal (%) = (Input-Output)/Input. Negative removal is percent increase in load after movement of runoff through filter strip.

TABLE 6.
Effects of different size buffer zones on reductions of sediment and nutrients from field surface runoff.

		Sediment			Nitrogen			Phosphorus		
Buffer Width	Buffer Type	Input. Conc.	Output Conc.	Reduction[4]	Input Conc.	Output Conc.	Reduction[4]	Input Conc.	Output Conc.	Reduction[4]
m		-- mg L^{-1} --		%	-- mg L^{-1} --		%	-- mg L^{-1} --		%
4.6[1]	Grass	7284	2841	61.0	14.11	13.55	4.0	11.30	8.09	28.5
9.2[1]	Grass	7284	1852	74.6	14.11	10.91	22.7	11.30	8.56	24.2
19.0[2,3]	Forest	6480	661	89.8	27.59	7.08	74.3	5.03	1.51	70.0
23.6[5]	Grass/ Forest	7284	290	96.0	14.11	3.48	75.3	11.30	2.43	78.5
28.2[6]	Grass/ Forest	7284	188	97.4	14.11	2.80	80.1	11.30	2.57	77.2

[1]Calculated from masses of total suspended solids, total N, total P, runoff depth, and plot size (22 x 55 m) from Magette et al. (1989)
[2]Input concentrations from Table 2, Peterjohn & Correll (1984). Nitrogen = Nitrate-N + exch. part. ammonium + diss. ammonium + part. organic N + diss. organic N. Phosphorus = part. P + diss P.
[3]Surface runoff concentrations at 19 m into forest reported by Peterjohn & Correll (1984). N and P constituents same as input (footnote 2).
[4]Percent reduction = 100 * (Input-Output)/Input.
[5]4.6 m grass buffer plus 19 m of forest.
[6]9.2 m grass buffer plus 19 m of forest.

iment concentration reduction found in Peterjohn and Correll (1984) to the output from a 4.6 m grass buffer (2841 mg L^{-1}) yields a sediment concentration of 290 mg L^{-1} from the 4.6 m grass and 19 m of forest (Table 6). This is an overall reduction of 96%. Applying the same approach to total N and total P yields an output concentrations of 3.48 and 2.43 mg L^{-1}, respectively. These represent concentration reductions of 75.3% and 78.5% for total N and total P, respectively. Increasing the width of the grass buffer to 9.2 m would increase sediment retention by 1.4% of input, N retention by 4.8%, but increase P concentrations slightly.

Although a number of experimental studies are ongoing which link grass filters and riparian forests for sediment and nutrient removal from surface runoff, most have only made preliminary reports. Parsons et al. (1994) reports sediment load reductions of 80 to 90% of field edge loads for both 4.2 and 8.5 m Ky-31 fescue buffer strips at a lower Coastal Plain site in North Carolina. Cutover riparian forests (perhaps analogous to early natural regeneration in Zone 2) showed somewhat higher sediment and total N yields than the 8.5 m grass strips. In a study of Zone 2 management on a tributary of Little R., Georgia, sediment loads in surface runoff entering the stream channel system were significantly higher from a clear cut Zone 2 than from a mature or thinned Zone 2 (Lowrance et al., unpublished). Although these results are preliminary they suggest the importance of the GVFS in Zone 3 during the early regeneration of Zone 2 after tree harvest. In a study of a reestablished RFBS, Vellidis et al.(1993), reports consistent but relatively minor reductions in PO_4-P in surface runoff in the first year after establishment of slash pine in a restored riparian forest buffer system in Little R. watershed.

3. Conclusions

For purposes of estimating riparian ecosystem functions in other physiographic regions, results from Inner Coastal Plain RFBS probably represent the upper limits for NPS pollution control in naturally occurring riparian forests equivalent to Zones 2 and 1. Other naturally occurring Coastal Plain and non-Coastal Plain systems are likely to be less effective than Inner Coastal Plain RFBS because of groundwater flow paths that bypass the riparian zone. Although numerous questions remain, the understanding of Coastal Plain riparian systems is much advanced compared to other portions of the CBW.

The ratio of source areas to RFBS which is required for continued improvement in water quality can probably be greatest in ICP conditions. Under optimum hydrologic conditions, such as the ICP, where groundwater moves in shallow pathways through naturally occurring riparian forests, a ratio of 2:1 or 3:1 (upland to riparian) is typical. These are the types of systems where some of the first data linking riparian forests and water quality were collected. However, data on nitrate concentration reductions suggest that much of the removal occurs within a relatively narrow ecotone at the field edge, implying that the ratio of field/forest can be increased. Management of upland source areas, to reduce NPS pollutants, and of RFBS, to increase effectiveness of removal of NPS pollutants, should provide opportunities for raising the ratio of cropland to RFBS.

Ongoing research on managed and experimental RFBS in the Coastal Plain suggests that restoration of the NPS pollution control function can be rapid, especially when nitrate moves through relict wetland soils. These studies also confirm the need to control channelized flow and to use an effective GVFS for sediment control when Zone 2 trees are harvested.

B. PIEDMONT

1. General Land Use and Hydrology

The Piedmont Province is an upland region lying between the Coastal Plain and the Valley and Ridge Provinces at elevations ranging from 30 to 300 m. The Piedmont accounts for 23% of the Chesapeake Drainage or 32,600 km^2 (NCRI Chesapeake, 1982). Of this area, 49% is in woodland, 25% is used as cropland, 4% is wetland, and 21% is in other uses including pastures, and suburban and urban land uses (NCRI Chesapeake, 1982). Of the total cropland within the Chesapeake drainage, 25% lies within the Piedmont.

The Piedmont is underlain primarily by metamorphic Precambrian and early Paleozoic rocks subject to several episodes of folding. The majority of Piedmont basement materials are quartzites, gneisses, schists, and marbles. These rocks were metamorphosed from ancient sandstones, gabbros and granites, shales, and limestones, respectively. During the Paleozoic, these basement rocks were interspersed with igneous pegmatite intrusions, and portions were covered by sedimentary deposits during the Triassic era. In Pennsylvania and Maryland, the marble belts form valleys; the gneiss, schist, quartzite, and granites form uplands (Hunt, 1974). Pavich et al., 1989 described the upland residual mantle (regolith) of Fairfax Co., VA as representative of the outer Piedmont Crystalline Province of Virginia and Maryland (Thornbury, 1965). The area has a high drainage density with most of its perennial streams incised into unweathered bedrock.

Given the great age of the rocks, the high degree of weathering, and absence of quaternary glaciation, the regolith (weathered rock, saprolite, subsoil, and soil) in the Piedmont can be quite deep. The maximum thickness of regolith is beneath flat upland hilltops. On schist, gneiss, and granite it is typically 15 to 30 m deep. Rocks such as serpentine and quartzite which weather slowly have thin regolith (Pavich et al., 1989). Throughout the outer Piedmont Crystalline Province, unweathered bedrock crops out in streams and regolith is generally thin or absent in valleys of perennial streams (Pavich et al., 1989). The contact between weathered and unweathered rock can be estimated on the side slopes of valleys by the location of heads of perennial streams at minor springs. Groundwater drains along the contact between weathered and unweathered rock and enters surface flow through springs (Pavich et al., 1989). Most of the groundwater storage in the Piedmont is within the regolith above the unweathered bedrock (Pavich et al., 1989). The saprolite acts as a relatively porous reservoir for groundwater. To a large extent, the depth of the regolith controls the hydrology of most Piedmont areas.

Based on hydrograph separations in the Piedmont of Chester County, Pennsylvania, Sloto (1994) found that baseflow ranges from 57 to 66% of watershed discharge, similar to estimates for the Virginia Piedmont of 60% (Pavich et al., 1989). The remainder of streamflow occurs during and following storms, but the proportion that is surface runoff, as opposed to enhanced subsurface flows (e.g., through near-stream rise in groundwater, drainage from soil layers, or rapid lateral transport through macropores), is difficult to determine. In forested watersheds, very little surface runoff occurs except from near-stream zones of high soil moisture. However, cultivated fields in the Piedmont generate greater surface runoff than fields in lower gradient Coastal Plains.

2. Control of Nonpoint Source Pollutants

Direct studies of NPS pollution control by RFBS have generally begun since 1990 so most results are preliminary at this point. The complex hydrogeology of the Piedmont Province will make generalizations from ongoing studies difficult even when final results are available. Most of the discussion to follow will focus on preliminary results from the North Carolina Piedmont which are most applicable to the southern portion of the Piedmont in the Bay watershed. Discussions of the geohydrology of the Piedmont and recent studies of the sources of water reaching streamflow will also be used to make inferences about the roles of RFBS in this province.

a. Removal of Nitrate from Groundwater

Groundwater in the Outer Piedmont Crystalline Province drains along the contact between weathered and unweathered rock and discharges through springs (Pavich et al., 1989). There are thought to be three pathways for groundwater discharge. In valleys underlain by weathered saprolite (often near headwaters), flow through the saprolite dominates baseflow. Water in the flow system is often oxic and may discharge nitrate directly to the stream channel. In valleys where streams have cut through the regolith to bedrock, springs begin in the valley flanks. Where streams have eroded to bedrock, discharge from fractures in the bedrock also contribute to streamflow. Stream discharge from the bedrock groundwater system is bypassed by the shallower systems if the regolith is not entirely eroded away. Even where bedrock contributes, most of the water in streams originates in the regolith because the volume of water in storage is so much greater than in the fractured bedrock.

Most groundwater recharge in the marble valleys occurs rather rapidly into fracture zones close to the land surface. The regolith, although variable in thickness, is usually thin and discharge to streams is probably from discrete fracture zones (in springs or directly into stream channels). As a result, there is probably little interaction of the groundwater with the root zone of riparian systems in the marble valleys.

In one study of nitrate transport in the Maryland Piedmont, McFarland (in press) found that streams contained nitrate concentrations of 5 to 10 mg NO_3-N L^{-1}. Most of the nitrate was attributed to discharge of water that was 0 to 5 years old from springs and from shallow flow systems in the regolith. Water in the bedrock was 20 to 30 years old with low or zero nitrate concentrations. Denitrification was suspected along older flow paths because of low dissolved oxygen and high iron concentrations in the water. This study indicates that riparian systems with deeply rooted vegetation may reduce nitrate in streams by removing nitrate from spring flow and the shallow flow systems through the regolith.

The only experimental study from the Piedmont that addresses the effectiveness of riparian buffers in mitigating subsurface flows of nonpoint pollutants is that of Daniels and Gilliam (in press), in which spatial and temporal patterns of groundwater nitrate at three sites in North Carolina were examined. Cultivated fields were separated from ephemeral or intermittent stream channels by 3 to 20 m of grass and naturally forested riparian buffers. Nitrate concentrations in groundwater under the cultivated fields exceeded 10 mg L^{-1}, but declined to lower levels in downslope wells. At one site, concentrations declined by as much as 30 mg L^{-1} over a distance of 20 m from the field edge. The study did not include mass balance analyses of nitrogen losses, and interpretation is complicated by the fact that streamflow nitrate concentrations exceeded those in near-stream wells. Thus, the authors were unable to partition actual nitrogen removal within the riparian zone from mixing (or dilution) effects, although they speculated that both were involved.

The results of Daniels and Gilliam (in press) are consistent with findings from Coastal Plain studies showing that high rates of nitrogen removal occur in areas with high water table conditions and shallow groundwater movement near the root zone. This suggests that the effectiveness of RFBS at particular sites throughout the Piedmont will depend strongly on the flowpaths of subsurface water in and near the riparian zone. Whatever the outcome of additional site-specific studies, it seems likely that regional estimates of RFBS effectiveness will also require data regarding hydrologic properties of near-stream zones.

The Piedmont is topographically diverse. In areas of gentle slopes and broad alluvial floodplains, the depth to groundwater in near-stream areas is probably 1 to 2 m as was the case in the North Carolina Piedmont study. In such areas there may be ample intersection of the saturated zone with the root system of riparian vegetation. This may also allow interaction between the saturated zone and soil layers containing sufficient organic matter to induce rapid denitrification. However in upland areas, the water table typically lies 3 to 10 m below the ground surface. In areas of steep terrain, it is common for smaller streams to be incised in relatively steep valleys. Near the fall line, larger streams also tend to have steep valley walls and minimal floodplains. Under these circumstances, the area of riparian forest in which the soils and root zone intersect the water table may be quite small.

Perhaps equally as important as water table elevation is depth or thickness of the aquifer in the near stream zone. While the bulk of subsurface water storage in the Piedmont occurs in the regolith, which may vary in depth from less than one meter to approximately 30 m, substantial storage occurs within a deeper zone of unweathered but fractured bedrock. The depth of the fractured zone, as indicated by well-water yields, may range roughly from 60 to 200 m, depending on rock type (Sloto, 1994). The likelihood that water reaches streams via shallow pathways, therefore, would depend both on the depth of the regolith in the vicinity of the stream, and on the proportionate contributions to streamflow from the regolith and from the fractured zone. Olmstead and Healy (1962) concluded from analyses of temporal patterns in baseflow and water table elevations in the Brandywine Valley of Pennsylvania that most streamflow originated from the regolith. Rose (1992, 1993) reached a similar conclusion from analyses of tritium variations in streamwater and groundwater in the Georgia Piedmont. If the regolith is beneath alluvial deposits near streams, much of the water reaching streams may pass through the riparian zone at substantial depths.

Another aspect of subsurface water movement that may prove important to RFBS effectiveness is the potential for lateral flow through near-surface soil layers. Lateral downslope water flow through unsaturated or briefly saturated soils may occur through macropores (Bevin and Germann, 1982) or where vertical flow is impeded by a soil horizon of low permeability (Gaskin et al., 1989; Schoeneberger and Amoozagar, 1990). There has been considerable investigation of shallow lateral drainage in other regions (e.g., Mosley, 1982; Mulholland et al., 1990; McDonnell, 1990) but only a few studies from the Piedmont Province.

Hooper et al. (1990) used end-member-mixing analysis (EMMA) of water chemistry to distinguish water sources in a Georgia Piedmont watershed. Their model used alkalinity, sulfate, sodium, magnesium, calcium and dissolved silica to identify three water sources: an organic soil horizon, hillslope drainage through subsoil and saprolite, and groundwater in bedrock. They concluded that hillslope drainage contributed a large portion of both baseflow and stormflow drainage during the wet winter months. Groundwater dominated the baseflow during the dry summer months with significant contributions from the organic horizon during storms. Comparable results were obtained by Rose (1992, 1993) in another study in the Georgia Piedmont. Rose inferred from analyses of tritium and inorganic analyses, that while baseflow during dry summer months originated from groundwater with an average residence time of 15 to 30 years, higher winter baseflows included a substantial component of water with a much shorter residence time (less than 10 yr) and lacking the chemical signature of groundwater.

In the North Carolina Piedmont, Daniels and Gilliam (in press) noted that soil water in an alluvium overlying saprolite was chemically distinct and apparently isolated from the deeper groundwater in the saprolite. They attributed the isolation to low permeability of the Bt soil horizon (subsoil compacted by tillage), and inferred that water in the soil layers traveled laterally above the Bt horizon into the riparian zone. Kaplan and Newbold (1993) hypothesized extended periods of soil water drainage following storms to explain patterns of dissolved organic carbon concentrations in a Pennsylvania Piedmont stream. The Bt horizon is well developed in Typic Hapludalfs and Typic Hapludults, soil groups which are common throughout the Piedmont, particularly in agricultural areas. In and near riparian zones, Aquic Fragiudults, are common. The fragipan associated with the latter soils probably also promotes lateral flow.

b. Removal of Sediment and Nutrients in Surface Runoff

The ability of RFBS to reduce nonpoint-source pollutants in overland flow may be of greater significance in the Piedmont than in the Coastal Plain because the steeper topography promotes greater velocities of overland flow. Daniels and Gilliam (in press) studied sediment and chemical reduction by GVFS and riparian areas for two years at six sites in the North Carolina Piedmont. They reported that the total sediment load reduction by the vegetated buffers during the study period ranged from 30 to 60 percent. However much of the sediment (mostly sand) passed through the vegetated buffers during one storm. When the results of that one storm were omitted from the calculations, the buffers removed approximately 80 percent of the sediment. Removals of silt plus clay averaged approximately 80% for the two-year study period. Total P removals in the filters ranged from 50 to 70%. Soluble orthophosphate removal was highly variable and usually was 50% or less. Removal of various forms of N was also variable and generally ranged from 40 to 60%. Most of the reductions were observed within 7 m of the field edge. The authors noted that the slope of the GVFS was less than that of the fields, so some of the sediment removal could be attributed to the change in slope alone. They further cautioned that the effectiveness of GVFS on steeper slopes might be limited. Where runoff from fields was directed as concentrated flow into riparian areas without complete vegetative cover, little or no reductions in either sediment or nutrients were observed. From these observations, Daniels and Gilliam (in press) recommended upslope dispersal of drainage water directed into forested areas.

Parsons et al. (1994a, b) conducted plot-scale studies in the North Carolina Piedmont on sediment and nutrient removals in grass and forest vegetated filters. They used 4 and 8 m grass buffers and 4 and 8 m forested filters to determine removal efficiencies. To date, they have monitored 50 storms over a three year period. They have observed that grass filters were somewhat more effective for sediment removal than the forest filters because of greater tendency for channelization in the forested area. Comparison of the grass and forest buffers is difficult because slopes were 4 to 6% in the grass filters as compared to slopes of 12 to 16% in the natural forested area. There was generally more ground cover on the grassed plots than in the forest, especially after grasses were reestablished in grass buffers.

Preliminary data from these studies are available for a maximum of four storms in 1991 (Parsons et al. 1994b). They found reduction of field edge sediment loads was consistently over 90% for both 4.3 m and 8.5 m forest buffers. Sediment loads were reduced 94% in the 4.3 m forest buffer (three storms) and 92% in the 8.5 m buffer (two storms). Reductions of nitrate, total Kjeldahl N, ortho-P and total P were more variable in these riparian forest buffers in the four 1991 storms. Although data are not available for all storms, it appears that the tendency to have channelized flow through the riparian forest area caused the high variation. For the available storm data lumped together, nitrate was reduced 41% compared to edge of field load for the 4.3 m forest buffer (four storms) and reduced 63 % in the 8.5 m buffer (two storms). Total kjeldahl N was reduced 67% in the 4.3 m buffer (three storms) but increased 14% in the one storm monitored in the 8.5 m buffer. Ortho-P, all of which was dissolved, decreased 6% in the 4.3 m forest buffer (three storms) but increased 17% in one storm through the 8.5 m forest buffer. Total P (sediment-bound + dissolved) decreased 50% in the 4.3 m buffer (three storms) but only 17% in the 8.5 m buffer (one storm). Although these data are preliminary, they show similar trends as some of the Coastal Plain runoff data with good control of sediment and sediment associated P but variable control of dissolved nutrients in surface runoff, especially dissolved P. More complete data from these studies should help guide design of RFBS for Piedmont landscapes.

3. Conclusions

Limited data from riparian forest studies in the Piedmont makes quantitative estimates of the NPS pollution control functions difficult. Patterns similar to ICP results seem to be present in studies from the North Carolina Piedmont with good control of nitrate in shallow flow paths and good control of sediment and sediment-borne pollutants in surface runoff.

Knowledge of the hydrology of certain parts of the Piedmont, such as the marble valleys of Pennsylvania and Maryland indicate a minor role for RFBS in control of groundwater borne pollutants. On smaller streams and in areas with thinner regolith, it appears that shallow groundwater movement through the subsoil and saprolite may be affected by RFBS. Buffer systems in the headwaters of streams where springs enter surface runoff may be effective, especially if the RFBS promotes the eventual presence of high organic matter soils in the areas of springs and permanent groundwater seeps. RFBS probably also intercept water moving in relatively shallow flow paths above texture discontinuities which promote lateral movement in the soil and subsoil. If extended periods of soils drainage above these texture discontinuities does occur, these waters should be subject to nutrient removal rates in RFBS similar to those in the ICP situations.

Although data are also limited on effects of RFBS on surface runoff, preliminary results indicate that the slope of RFBS may limit effectiveness because of channelization through forests. Relatively steep RFBS will certainly benefit from the presence of a well managed Zone 3 at the field edge and may require level-lipped spreaders to control the tendency of surface runoff to create permanent channels. In reestablished areas on relatively steep slopes, such as the 12 to 16% slopes reported on from North Carolina, a high stocking density of trees in Zone 2 is warranted. This would have the effect of both increasing resistance to surface flow by increased numbers of stems as well as providing a high level of root biomass more quickly than lower stand densities.

C. VALLEY AND RIDGE

1. General Land Use and Hydrology

Valley and Ridge physiographic province is the area in which structures due to folding dominate the topography. The Valley and Ridge and Appalachian Plateau make up about 60% of the CBW (Table 1). Geomorphologically, the Valley and Ridge province is one of folded mountains in which resistant strata form ridges and weaker rocks are worn down to lowlands. Valleys within this province are underlain by limestone or shale and the ridges are capped by the more resistant rocks (well-cemented siliceous sandstone and conglomerate).

The physical characteristics of this province are intimately connected with its streams which are primarily causes of the present topography. Streams develop mostly on belts of soft rock crossing hard rock ridges infrequently and usually at right angles. The Bay watershed encompasses the middle section of the Valley and Ridge. Distinctive features of this section are conspicuous trellised drainage patterns and a comparative absence of ridges on its southeastern one-quarter to one-third, the Great Valley (Fenneman, 1938).

Heath (1984) placed the Valley and Ridge in the Central Nonglaciated groundwater region. The region is characterized by thin regolith underlain by fractured sedimentary bedrock. The principal water-bearing openings in the bedrock are fractures which develop both along bedding planes and across them at steep angles. Openings developed along the fractures are usually less than 1 mm wide. The principal exception to this is in limestone, where water moving through the original fractures has enlarged them to form, at the extreme, extensive cavernous systems capable of transmitting large amounts of subsurface flow. Recharge of groundwater in this region generally occurs in outcrop areas of the bedrock aquifers in the uplands between streams. Discharge from the groundwater system is by springs, seepage areas, and direct inflow to the stream bed, and by evaporation and transpiration in the near-stream areas where the water table is near the land surface.

The aquifers in the Valley and Ridge are unconfined with little matrix permeability and low storage coefficients. Groundwater circulation is limited at depths greater than 100 m due to the decrease in fracture size and frequency. Even though the entire Valley and Ridge falls within the Central Nonglaciated groundwater region, there are substantial differences in flow characteristics between the limestone and shale valleys, and among the limestone valleys. Flow characteristics are most complicated within the limestone aquifers and connections between lower-order streams and regional groundwater are quite variable in time and space.

2. Control of Nonpoint Source Pollution

Despite the Valley and Ridge and Appalachian Plateau comprising a large portion of the CBW, only a small number of research projects have been conducted to evaluate the amelioration of NPS pollution in riparian buffers within this area. These projects addressed within-stream water quality (e.g., cold water fisheries habitat, macroinvertebrates, and sedimentation) and Bay-scale water quality (export of plant nutrients, pesticides and suspended solids).

The entire study of riparian ecosystems relative to stream quality was begun within the last 20 years. That little of it was conducted in the Valley and Ridge may be explained by the small amount of wetlands in this province. The conditions which promote improvements in the chemical composition of waters discharging through riparian ecosystems (small land surface slopes, high water tables and low aeration status) are commonly associated with wetlands. Only 1% of the Valley and Ridge located in the CBW is classified as wetlands, which constitutes 7% of the wetlands in the CBW (Table 1). This contrasts with the 57% of CBW wetlands on the Coastal Plain comprising 21% of the Coastal Plain within the CBW.

The most intensive agricultural NPS pollution occurs in the limestone valleys of the Valley and Ridge.

Study of subsurface hydrology necessary to determine the extent of groundwater renovation in the riparian zone is difficult and expensive in karstic limestones. A paucity of studies in the Valley and Ridge may be due to the major source of NPS pollutants being located over an aquifer with complex hydrology for the scale of riparian zone studies.

The processes which renovate surface and groundwater within riparian ecosystems are the same in the Valley and Ridge as in the other physiographic provinces and much can be inferred from research done in the Atlantic Coastal Plain. When watershed morphology and aquifer characteristics are compared (Schnabel et al , 1994), general statements can be made about the likelihood of ground and surface water renovation in riparian zones of the Valley and Ridge relative to the Coastal Plain. However, research conducted within the Valley and Ridge must be evaluated to quantify the impact of riparian buffers on stream hydrology, chemistry and biology.

a. Removal of Nitrate from Groundwater

The Mahantango Creek Watershed, a USDA-ARS research watershed, is located within the Susquehanna R. Basin approximately 40 km north of Harrisburg, Pennsylvania. Topography, geology, and land use of the Mahantango Creek Watershed are typical of upland watersheds in the unglaciated, intensely folded and faulted Valley and Ridge Province. These watersheds generally have relatively steep land-surface slopes and minimal floodplain development or alluvium. Most stream reaches expose bedrock over all or part of their length. Land use within the watershed is approximately 57% cropland, 35% forest and woodlots, and 8% permanent pasture. Elevation ranges from 240 to 480 m msl. The northern ridge is covered with a mature deciduous forest, while agricultural land use predominates in the remainder of the watershed. Climate is humid and temperate, and rainfall averages about 1150 mm yr^{-1}.

Groundwater provides most of the 60 to 80% of streamflow estimated to be subsurface return flow (Gburek et al., 1986). Primary recharge occurs in the late fall, winter, and early spring months, but minor recharge can occur during the growing season following large single or grouped precipitation events. Because ridge-top soils are highly permeable, nearly all rainfall infiltrates. In contrast, the finer-textured poorly drained soils adjacent to the stream often function as groundwater discharge zones during the dormant season.

The Mahantango Creek Watershed is underlain by two geologic formations, Trimmers Rock (Late Devonian) and Catskill (Late Devonian—Early Mississippian). Previous analysis of well yields indicated that rock fracture patterns are as important to formation permeability as rock type, and based on specific capacity data (Urban, 1977), the two formations are hydrologically similar. A shallow, approximately 10 to 15 m layer of weathered fractured bedrock overlays the entire watershed and has hydraulic properties different from those of the deeper, less-fractured layer (Gburek and Urban, 1990). The two-layer aquifer, with its upper, highly transmissive layer, permits rapid horizontal groundwater throughflow while also leaking to recharge the deeper layer. Differing land uses in the area recharging groundwater, the layered subsurface permeability distribution, and the general pattern of groundwater flow are expected to result in a general pattern of higher nitrate concentration in shallow groundwater and lower concentration in the deep groundwater. In the experimental area, all aquifer waters, both shallow and deep, discharge to the surface streams.

Although it is a very small portion of the watershed area, the near-stream zone exerts major controls on stream flow chemistry and hydrology. Because it is hydrologically dynamic, particularly as related to seep zone formation, the near-stream zone can control the amount and timing of surface runoff and, thus downstream flooding. The water table response to storms strongly influences or controls subsurface discharge, the nature and extent of riparian vegetation, stream bank stability, and the nature of the chemical and biological systems to which chemicals in the discharge are exposed.

Nitrogen and phosphorus species were measured in surface runoff and seepage waters in a grassed buffer between a first-order stream and a cropped field, during and immediately after storms to determine how surface and subsurface waters interact to generate streamflow during storms (Pionke et al., 1988). Nitrate concentration in seepage and base flow were similar and typically exceeded concentrations in surface runoff, rainfall, and peak storm flow by 5 to 20 times. The median nitrate concentration observed in seepage was similar to mean concentration observed in stream base flow at the outlet to a 9.9-ha catchment over a 2-yr period and similar to those computed from a hydrologic and nitrogen mass balance for agricultural groundwaters of Mahantango Creek Watershed (Pionke and Urban, 1984). They concluded that hydrologic conditions mainly determined nitrate concentration and load delivered to the stream over the short term, as hydrology affected both the surface runoff:subsurface discharge ratio and the volume of subsurface discharge. This indicates that surface-groundwater interactions are more frequent and longer lasting on lower portions of the watersheds. Consequently, in this and similar watersheds the nitrate content of groundwater is less likely to be altered in the riparian zone in the upper portions of head water streams.

Another study on Mahantango Creek focused on the role of existing riparian zones. A strip of woods (20-60 m) was located between the stream and cropland on both sides at the study site. There was a break in slope on one side of the stream where the land surface flattens as it approached the stream. The woods were removed from this flatter area bordering the stream 15 to 20 years before the study and it was seeded to grass. Thus the vegetation pattern moving up-gradient on one side of the stream was a relatively flat, well-maintained grass strip, a steeper strip of woods and then cropland. On the other side a steep strip of woods separated the stream from cropland. Nitrate-N concentrations in shallow groundwater under the grass strip were reduced by 25 to 50% between 9 m and 6 m from the stream during the growing season. There were generally small differences in nitrate-N concentrations in shallow groundwater 3 m from the stream and baseflow in the stream. The water table was frequently deeper than 1 m, particularly on the wooded side of the stream. The wooded side was much steeper and didn't develop seepage zones as frequently as on the less steep grassed side of the stream (Schnabel, 1986). A pattern similar to the nitrate concentrations measured in the grassed riparian zone was found in deeper groundwater (3 m) beneath the wooded riparian zone. Gburek et al. (1986) estimated that nitrate reduction within the riparian zone of the Mahantango Creek Watershed was equivalent to only 4% of the mineral N exported from the watershed during the year. The limited impact of riparian processes on total N export resulted from the small area near the stream thought to support denitrification at optimum rates, combined with the fact that the area generally expands after soil temperatures begin to decrease, presumably limiting denitrification and plant uptake rates.

The chemical composition of the aquifer differs with depth. While recharge for the deeper part of the aquifer originates at the wooded ridge tops, the shallower part of the aquifer is recharged in the agricultural interior of the watershed. From simulation with a mixing model which viewed baseflow as a mixture of discharge from the shallow fractured part of the aquifer and deeper, less fractured portion of the aquifer, Schnabel et al. (1993) concluded that the riparian zone was not the major control on temporal variation in nitrate concentration at the outlet to Mahantango Creek Watershed.

A study designed to examine groundwater nitrate dynamics was conducted in the western portion of the Valley and Ridge Province. Altman and Parizek (1994) conducted a study of nitrate movement from a field through the riparian zone of a tributary of Bald Eagle Creek at the western edge of the Valley and Ridge in Pennsylvania. They found that nitrate levels in groundwater decreased from 5 to 8 mg NO_3-N L^{-1} beneath the field to less than 0.5 mg NO_3-N L^{-1} in the riparian zone. Based on flow-net analysis, they concluded that water sampled in the riparian zone apparently did not originate from the crop area with elevated nitrate levels. The groundwater flow direction did not follow the surface topography but instead followed the local bedrock topography. Groundwater was actually flowing toward the larger creek which the tributary stream was feeding. Their report did not address the fate of the nitrate enriched water as it moved through the riparian system associated with Bald Eagle Creek. This study does point out the difficulty of research on groundwater and associated solute movement in areas of complex hydrogeology such as the Valley and Ridge.

b. Streamflow Transport of Phosphorus

Sediment and water associated phosphorus export from the Mahantango Creek Watershed and a 9.9-ha subcatchment was examined for a 4-yr period to determine the mode of phosphorus transport from a typical Valley and Ridge upland watershed (Pionke and Kunishi, 1992). During storms, most of the labile P (sum of total soluble P and sediment P extracted by Cl resin) was exported from Mahantango Creek Watershed in the dissolved rather than the particulate phase. The dissolved P dominated because the dilution of sediment by runoff (~3000:1) more than compensated for the greater concentration of labile P compared to soluble P concentration (~1000:1). In contrast, storm flow transport of algal-available (sediment P extracted by 0.1 N NaOH) total P was mostly with sediment, largely because concentrations of both on sediment greatly exceeded labile sediment P concentrations. When combined with P concentrations in base flow, which accounts for approximately 80% of total flow, 50 and 28% of algal-available and total P, respectively, were exported from Mahantango Creek Watershed in the dissolved phase. Thus, the most readily available P components are transported in the dissolved rather than particulate forms. This has important implications for use of RFBS to control P losses from agricultural land. If RFBS are more effective at controlling particulate P than dissolved P, higher proportions of P in the dissolved phase would imply less effective overall control of P transport.

c. Removal of Sediment and Nutrients in Surface Runoff

Studies by Dillaha et al. (1988, 1989a,b) have shown the potential efficacy and limitations of grassed filter strips for controlling NPS pollution. Near Blacksburg, Virginia, Dillaha et al. (1988) studied the use of orchardgrass (*Dactylis glomerata*) GVFS for controlling potential sediment and nutrient losses from feedlots. Plots received 7,500 and 15,000 kg ha^{-1} of fresh dairy manure and had slopes of 11 and 16%. In plots with shallow, uniform surface flow, 81 and 91% of the sediment and soluble solids were removed by 4.6 and 9.2 m GVFS, respectively. In plots where concentrated flow was allowed to occur, removal was much less. The GVFS were ineffective for controlling dissolved nutrients and nutrients associated with fine sediment. Concentrations of soluble N and P in effluents from GVFS were found to be high enough to cause eutrophication in receiving waters. Concentrations of soluble inorganic N were as high as 8.2 and 5.1 mg N L^{-1} from the 4.6 and 9.2 m GVFS, respectively.

In a similar study of orchardgrass filter strips below fertilized cropland, Dillaha et al. (1989b) obtained comparable results to the feedlot experiment. The sediment was initially trapped at the top of the GVFS. However, the GVFS became ineffective as it gradually became inundated with sediment.

In surveys of farms that employed GVFS along streams in Virginia, Dillaha et al., 1989a,b) found that in Valley and Ridge areas, the GVFS tended to be less effective than in flatter Coastal Plain sites. Except for localized erosion control along the stream bank, GVFS did little to mitigate NPS pollution from the upland in the Valley and Ridge because surface runoff usually became concentrated within the fields in natural drainageways before entering the GVFS. In general, the GVFS were most effective below smaller fields where water could enter the GVFS before it had an opportunity to concentrate.

Even where the GVFS had potential for sediment trapping, in many cases inadequate maintenance had rendered them ineffective (Dillaha et al., 1989a). Lack of mowing sometimes allowed taller weeds to shade out low ground cover, thereby reducing the capability of the GVFS to trap sediment. Erosion across the GVFS had caused severe gully problems in some cases. Heavy traffic had sometimes damaged the sod and created ruts. Sediment buildup on some sites had caused the upper margin of the GVFS to be higher than the adjacent field. Or sometimes, ditches from moldboard plowing were created parallel to the upper edge of the GVFS. In either of these two latter situations, water would run parallel to the edge of the GVFS until it could get across it in concentrated flow.

3. RFBS in Forested Watersheds

Although the RFBS is designed for use adjacent to agricultural areas, a number of forestry experiments in the Valley and Ridge Province provide useful general information on hydrology, sediment transport, and sustainability of riparian forest buffers. A series of experiments were begun in the late 60's and early 70's by Forest Service personnel to design BMPs for logging operations in response to the Federal Water Pollution Control Act Amendments of 1972 (P.L. 92-500). In many of these experiments, a strip of trees was left standing along perennial streams to protect the stream from NPS pollutants. The experimental sites included locations in Pennsylvania, West Virginia, and Tennessee.

The Leading Ridge Experimental Watershed Research Unit is located in the Ridge and Valley Province of central Pennsylvania and consists of three adjacent watersheds. BMPs used on a commercially clearcut watershed were designed to minimize stream sedimentation from silvicultural operations. These practices included maintaining a 30 m buffer strip on each side of perennial streams and restricting slash piles and log-landing sites from the vicinity of stream channels (Lynch and Corbett, 1990). A comparison of suspended sediment concentrations on the Leading Ridge Experimental Watersheds for the first two years after clearcutting shows that the BMPs were effective. Average suspended sediment concentrations of 1.7, 10.4 and 5.9 mg L^{-1} were reported for an uncut control watershed, a clearcut and herbicide-treated watershed without BMPs, and a commercially clearcut watershed with the riparian buffer strip, respectively, for the first year after treatment (Lynch et al., 1985; Lynch and Corbett, 1990). Water yield increased following all clearcutting treatments. The greatest increase, equivalent to 32 cm over the area cut, occurred the first year after clearcutting and herbicide treatment to control regrowth. Annual yield was sharply lower the second year and was not statistically different from water yield on the control watershed at the end of the fourth year following harvesting (Lynch and Corbett, 1990). Average suspended sediment concentrations in the clearcut/herbicide treatment jumped to 78.7 mg L^{-1} during the second year after treatment compared to 5.1 and 9.3 mg L^{-1} for the control watershed and clearcut watersheds with riparian buffers, respectively (Lynch et al., 1985; Lynch and Corbett, 1990). The extremely high sediment concentrations on the clearcut/herbicide treatment were attributed to channel cutting, and bank erosion and slumping on the lower portion of the channel. Increased sediment concentrations on the commercially clearcut watershed the second year (9.3 mg L^{-1} compared to 5.9 mg L^{-1}) were attributed to tree blowdown along a 460 m length of intermittent stream which did not have a buffer along it. The few remaining trees in the section blew over and loosened the soil. The increased water yield resulting from the clearcut caused the intermittent stream to flow, permitting the transport of soil to the stream channel. This illustrates the need for maintaining riparian buffer strips along intermittent streams (Lynch et al., 1985).

Average annual nitrate concentration in the clearcut/herbicide watershed was substantially increased compared to the control watershed (2.54 vs. 0.11 mg NO_3-N L^{-1}). Increased nitrate concentrations combined with increased water yields from the treated watershed resulted in significantly greater N loading than the control. However, rapid revegetation, which is almost impossible to prevent in the humid East, generally prevented any major stream enrichment problems. Where BMPs were used, nitrate concentrations were substantially less than the clearcut/herbicide treatment, although significantly higher than control (0.37 vs. 0.08 mg NO_3-N L^{-1}) for the first two years after clearcut (Lynch et al., 1985).

Mulholland et al. (1990) studied an area similar to the Valley and Ridge region of the Chesapeake Bay. They investigated the hydrogeochemical response of the West Fork of the Walker Branch Watershed in eastern Tennessee to four large storms. The study area was a 38.4-ha forested watershed with deep, highly weathered soils, a network of ephemeral stream channels, and a spring-fed perennial stream which flowed over dolomite bedrock in the lower portion of the watershed. The watershed has broad ridges which slope steeply to narrow valleys. Surface soils have very high hydraulic conductivities due to macroporosity associated with forest soil formation processes. Reduced hydraulic conductivities at depth in the soil are associated with increasing clay content. The weathered zone ranges in depth from a meter near the stream to about 30 m at the basin divides.

In this watershed, water held above the shallow restrictive layer flowed through the rhizosphere and was virtually depleted of nitrate. However, water passing through the restrictive layer (apparently the layer was quite leaky) had higher nitrate concentrations. Groundwater transferred between catchments or leaked to deeper groundwater and discharged near the watershed outlet bypassed the riparian zone closest to the source of NPS pollutants. Where these transfers occurred, groundwater was less likely to be renovated by riparian zone processes.

4. Conclusions

Forested riparian buffers have proven effective in controlling water temperature and sediment delivery to streams in forest and agricultural settings within the Valley and Ridge. Our knowledge of groundwater renovation in riparian ecosystems is less certain. Where regolith is thin and bedrock controls subsurface flow, seepage faces or springs produce relatively small saturated areas with wetland characteristics. Attenuation of nitrate concentrations may occur if RFBS are restored in these seepage areas. In contrast, where regolith is deep with flow-restrictive layers near the land surface, shallow flow systems develop on the confining layers resulting in more extensive riparian ecosystems where groundwater discharges to the streams. These conditions, more likely in the glaciated Appalachian Plateau than in the Valley and Ridge, are likely to have higher overall nitrate removal rates.

III Applicability of the Three Zone Riparian Buffer System

The three-zone RFBS specification is based on studies of naturally occurring riparian forests along low order (1st to 4th order) streams and experimental scale grass filter strips. Under natural conditions, riparian forest ecosystems formed a dynamic yet stable buffering system along most shorelines, rivers and streams in the Bay watershed. Although few studies have documented the specific changes in water quality functions during the establishment period of a riparian forest, established RFBS are expected to sustain water quality functions over the long term in a manner similar to the natural system.

The effect of upstream activities which modify hydrology or pollutant loads, loading rates, or the change in functions due to management of the RFBS, such as timber harvest, add uncertainty and risk to predicting changes in some water quality functions over time. However, existing research, knowledge of riparian ecology, and experience with related hydrologic systems can form the basis for recommendations on the applicability of RFBS. The 12 member scientific panel that prepared this report utilized these resources to produce the following set of Best Professional Judgements (BPJ) of conditions and criteria for assessing the effectiveness of the three-zone RFBS for use in the CBW.

A. CONTROL OF THE STREAM ENVIRONMENT

Control of the stream environment will occur in almost all cases along smaller streams with Zone 1 vegetation. The environments of tidal streams, tidal portions of the bay, and larger rivers maybe controlled by other factors more than the immediate riparian ecosystem. The consensus BPJ are:

1) Control of the stream environment for aquatic ecosystems is most likely to be achieved with vegetation approximating the original native vegetation along streams.
2) Control of the stream environment will be affected less by physiographic regions than by size of stream. As the size of stream or water body increases, most effects of the riparian system on the stream environment decrease. However, the habitat functions of large woody debris are important even on large river banks and on Bay shorelines.
3) Just as Zone 1 may also play an important role in NPS pollution removal, Zone 2 may play an important supporting role in controlling the stream environment.

In many cases, especially along higher order streams, quality of the stream environment will reflect the influence of both Zone 1 and Zone 2. Sustainability of Zone 1 function may depend on proper management of Zone 2. Where windthrow of trees or stream bank stability is a problem, Zone 2 vegetation should be managed with long rotations, thinning cuts, or other practices which minimize the time and areal extent of a non-forest Zone 2. The general goal would be to minimize both the amount of time and the stream length for which Zone 1 would be the only riparian forest. Where Zone 1 will function alone, increased width and/or other adjustments may be required to enhance sustainability.

B. CONTROL OF NONPOINT SOURCE POLLUTION

Unlike the processes involved with control of the stream environment, the functions of riparian systems to control NPS pollution are dependent on hydrologic connection(s) of pollutant source(s) with the riparian forest buffer system. Although generalizations will be made, the extent, timing, and spatial variability of the hydrologic connections add uncertainty to BPJ assessment of NPS pollution control. The hydrologic connection between source areas and riparian ecosystems probably ranges from nearly 100% of the water moving across the surface or in shallow groundwater through the biologically active soil zones (e.g., ICP Watersheds) to a very low percentage of flow moving through riparian ecosystems. This lower limit is not well defined, but a conservative estimate can be made by hydrograph analysis to separate storm flow from baseflow. At a minimum, most stormflow should move in either surface runoff or shallow groundwater and should be subject to processing in a RFBS.

For either surface runoff or shallow groundwater, removal of NPS pollutants in RFBS is first determined by the hydrologic pathways and then modified by interactions of hydrology, soils, geochemical environments, management, loading, and vegetation. As pointed out above in Sections I and II, some of these factors are poorly understood and most are poorly quantified, especially outside the ICP from which much of the existing information is derived.

As a means of conceptualizing the NPS pollution control functions of riparian ecosystems in the CBW, a series of flow diagrams for different physiographic settings was developed (Fig. 6 through 14). These figures are generally representative of many of the different hydrologic settings within the regions and provide reference points for discussions (below) concerning hydrologic controls on the NPS removal function. It is important to note that these diagrams are generalized and that more than one hydrologic setting may be present in larger watersheds. The consensus BPJ decisions are summarized for nitrate removal, sediment and sediment-borne pollutant removal, and phosphorus (from all sources) removal in Tables 7 through 9.

1. Coastal Plain

a. Inner Coastal

The best information on RFBS comes from Coastal Plain systems represented by Figure 6. In these ICP systems, most of the excess precipitation moves to streams via subsurface runoff or shallow groundwater movement. Most or all of this water moves in or near the root zone or is subject to capillary transport due to transpiration from the root zone. The ICP represents one end of the spectrum of riparian ecosystems function for removal of NPS pollutants. In these systems, riparian ecosystems exert substantial control over both the hydrologic and nutrient transport response of agricultural watersheds. ICP areas, represented in Section II by Rhode R. in Maryland and areas in Georgia and North Carolina, are typically areas with a high density of stream channels, well developed "natural" riparian forests, and extensive connections between agricultural fields and riparian forest ecosystems. Most of the Western Shore and the upper Eastern Shore Coastal Plain in the CBW is considered ICP. Because of the relatively large amount of scientific data collected from ICP type systems, primarily in MD, NC, and GA (see Section II), the scientific panel was able to make the most comprehensive consensus BPJ for these areas. Among these conclusions are the following:

1) Based on mass balances, established RFBS remove 20 to 39 kg NO_3-N ha^{-1} yr^{-1} from subsurface flow.

2) Based on mass balances, total N retention in established systems ranges from 26 to 74 kg N ha^{-1} yr^{-1}.

3) For the RFBS to be applicable in systems with artificial drainage near streams, the drainage system will have to be modified to work in conjunction with the RFBS.

4) Newly established systems are likely to have a substantial effect on subsurface nitrate loads in (at most) 5 to 10 years if anoxic sediments and high organic matter surface soils are already in place. By 15 to 20 years, reestablished RFBS should control groundwater nitrate loads in most (if not all) ICP situations. Reestablishment of RFBS along all streams in the ICP is likely to lead to water quality improvements.

5) The nitrate concentration data from ICP systems indicates that higher nitrate loadings could be removed in the RFBS if it was exposed to higher loadings than represented in the mass balance studies. This is most likely to be true in systems with highest denitrification rates or potentials.

6) Based on the mass balances, net retention of phosphorus in established systems is 1.2 to 2.9 kg P ha^{-1} yr^{-1}. Retention of phosphorus in surface runoff appears to be mainly through retention of particulate phosphorus and infiltration in the RFBS. Retention of dissolved ortho-P appears to be considerably less effective for both surface runoff and subsurface flow.

TABLE 7.
Removal of nitrate from groundwater—summary

Hydrologic Setting	*Expected Level**	*Critical Constraints*	*Restoration/ Enhancement*	*Management Factors*
1. Inner Coastal Plain	High, most water moves in or near root zone.	Bypass flow due to subsurface drains.	Important on all streams. Rapid restoration of function by denitrification.	Maintain vegetation/ denitrification link.
2. Outer Coastal Plain -Well Drained upland.	Low, primarily removal from shorter flow paths.	Bypass flow due to deeper aquifers. Long flow paths surface in stream channels.	Concentrate on headwater areas. Zone 1 important for nitrate removal.	Manage vegetation in headwaters for deep rooting and N uptake.
3. Outer Coastal Plain -Poorly Drained Upland/Surficial Confined.	Medium/High	Lower loadings. Lower rates of removal in head- water areas.	Restore in headwater areas. Rapid restoration of function by denitrification.	Potential for higher removal with higher loadings due to anoxic soils. Denitrification on larger streams.
4. Tidal-Coastal Plain	Low/Medium	Depth to water-tables. Bank erosion due to unstable soils.	Limit practice to areas without marsh wetlands down slope.	Enhance vegetation uptake, link vegetation and denitrification.
5. Piedmont -thin soils/Triassic Shales	High	Lower loadings than ICP. Valley shapes control local flow paths	Select deeply rooted vegetation, restore small and large streams, seepage areas.	Infiltration of surface runoff, may effect nitrate processing.
6. Piedmont -Schist/Gneiss bedrock	Medium	More flow into regional aquifers, bypassing riparian zone.	Select deeply rooted vegetation. Restore in seepage areas.	Enhance vegetation uptake, link vegetation and denitrification.
7. Piedmont/Valley & Ridge - Marble/Limestone Bedrock	Low	Most flow into regional aquifers and into large rivers.	Select deeply rooted vegetation. Restore in seepage areas.	Enhance vegetation uptake, link vegetation and denitrification.
8. Valley & Ridge - Sandstone/Shale Bedrock	Medium/High	Presence of seeps and floodplains. Valley configurations.	Select deeply rooted vegetation. Restore in downstream areas and seepage areas.	Enhance vegetation uptake, especially early in growing season.
9. Valley & Ridge/ Appalachian - Low order streams	Medium/High	Residence time of water. Presence of seeps and flood plains.	Select deeply rooted vegetation. Restore floodplains and seepage areas.	Enhance vegetation uptake. Zone 1 may be more important for removal due to water nearing surface.

*Expected level of function based on mature RFBS. Where possible level is quantified in text. Expected levels are relative to outputs from agricultural lands in the hydrologic settings.

TABLE 8.
Removal of phosphorus from all sources—summary

Hydrologic Setting	*Expected Level**	*Critical Constraints*	*Restoration/ Enhancement*	*Management Factors*
1. Inner Coastal Plain	Medium/Low	Control of dissolved P in surface runoff and groundwater is limited.	Restore in areas with major P load in surface runoff. Enhance existing forest with Zone 3.	Increase uptake in biomass and P accretion in vegetation, litter & soil.
2. Outer Coastal Plain -Well Drained upland.	Medium/Low	Control of dissolved P in surface runoff and groundwater is limited.	Restore in areas with major P load in surface runoff. Enhance existing forest with Zone 3.	Increase uptake in biomass and P accretion in vegetation, litter & soil.
3. Outer Coastal Plain -Poorly Drained upland/surficial confined	Medium/Low	Control of dissolved P in surface runoff and groundwater is limited.	Restore in areas with major P load in surface runoff. Enhance existing forest with Zone 3.	Increase uptake in biomass and P accretion in vegetation, litter & soil.
4. Tidal-Coastal Plain	Medium/Low	Control of dissolved P in surface runoff and groundwater is limited.	Restore where P load is in surface runoff. Enhance existing forests and grass strips.	Increase uptake in biomass and P accretion in vegetation, litter & soil.
5. Piedmont -thin soils/Triassic Shales	Medium/Low	Control of dissolved P in surface runoff.	Restore in areas with large surface runoff P loads.	Increase infiltration and filtering of fine sediment.
6. Piedmont -Schist/Gneiss bedrock	Medium/Low	Control of dissolved P in surface runoff.	Restore in areas with large surface runoff P loads.	Increase infiltration and filtering of fine sediment.
7. Piedmont/ Valley & Ridge -Marble/Limestone Bedrock	Medium/Low	Control of dissolved P in surface runoff.	Restore in areas with large surface runoff P loads.	Increase infiltration and filtering of fine sediment.
8. Valley & Ridge -Sandstone Shale Bedrock	Medium/Low	Control of dissolved P in surface runoff.	Restore in areas with large surface runoff P loads.	Increase infiltration and filtering of fine sediment.
9. Valley & Ridge/ Appalachian -Low order streams	Medium/Low	Control of dissolved P in surface runoff.	Restore in areas with large surface runoff P loads.	Increase infiltration and filtering of fine sediment.

* Expected level of function based on mature RFBS. Where possible level is quantified in text. Expected levels are relative to outputs from agricultural lands in the hydrologic settings.

TABLE 9.
Removal of sediment and sediment-borne pollutants (including sediment-borne P)—summary

Hydrologic Setting	*Expected Level**	*Critical Constraints*	*Restoration/ Enhancement*	*Management Factors*
1. Inner Coastal Plain	High/Medium	Concentrated flow must be converted to sheet flow.	Restore in all areas. Enhance existing forest with Zone 3.	Maintain ground cover in Zone 3 & 2. Use spreaders in Zone 3.
2. Outer Coastal Plain -Well Drained upland	High/Medium	Concentrated flow must be converted to sheet flow.	On larger streams, focus on filtering eroded sediment.	Practices to increase resistance to failure of Zone 3 and 2.
3. Outer Coastal Plain -Poorly drained upland/Surficial confined	High/Medium	Less surface runoff but similar efficiencies as in other CP systems.	Enhance vegetation in broad existing areas. Restore in headwaters.	Manage for sediment removal in headwater streams.
4. Tidal-Coastal Plain	High/Medium	Convert concentrated flow to sheet flow. Bank stability limits usefulness in some areas.	Restore/enhance in all areas. Limit to wider Zone 3 in some areas.	Practices to increase resistance to failure of Zone 3.
5. Piedmont -thin soils/Triassic shales	High/Medium	Slope of non-floodplain areas. Volumes of surface runoff.	Restore in all areas. Function dependent on Zone 3 in first few years.	Increase management of Zone 3 to control coarse sediment and spread flow.
6. Piedmont -Schist/ Gneiss bedrock	High/Medium	Slope of non-floodplain areas. Sediment loads in stream flow from valley sides.	Restore in all areas with erosion impacting streams. Enhance existing forests with Zone 3.	Increase management of Zone 3 to control coarse sediment and spread flow.
7. Piedmont/Valley & Ridge - Marble/Limestone Bedrock	High/Medium	Slope of non-floodplain areas. Sediment loads in stream flow from valley sides.	Restore in all areas with erosion impacting streams. Enhance existing forests with Zone 3.	Increase management of Zone 3 to control coarse sediment and spread flow.
8. Valley & Ridge - Sandstone/Shale Bedrock	High/Medium	Sediment loads in stream flow from valley walls. Slopes of non-floodplains.	Restore in all areas with erosion impacting streams. Enhance existing forests with Zone 3.	Increase management of Zone 3 to control coarse sediment and spread flow.
9. Valley & Ridge Appalachian -Low order streams	High/Medium	Sediment loads in stream flow from valley walls. Slopes of non-floodplains.	Restore in all areas with erosion impacting streams. Enhance existing forests with Zone 3.	Increase management of Zone 3 to control coarse sediment and spread flow.

* Expected level of function is based on mature RFBS. Where possible level is quantified in text. Expected levels are relative to outputs of nitrate from agricultural lands in the hydrologic setting.

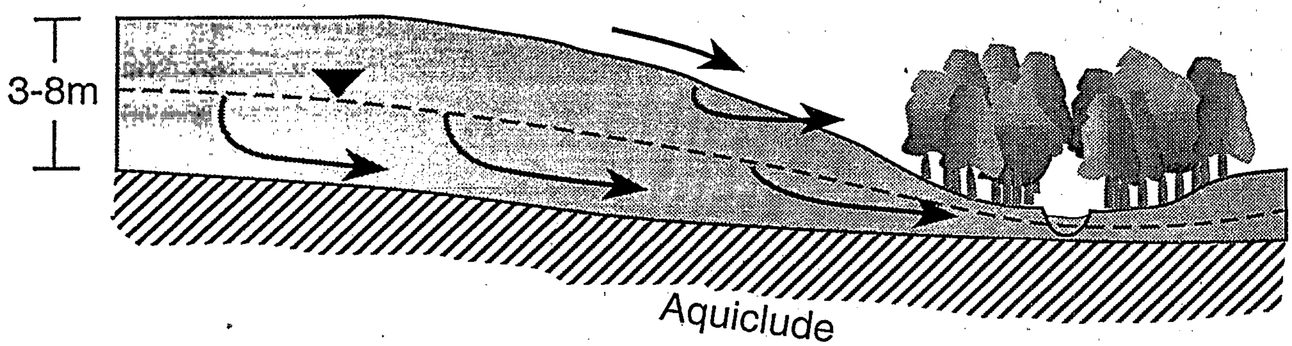

FIGURE 6. Inner Coastal Plain flow system.

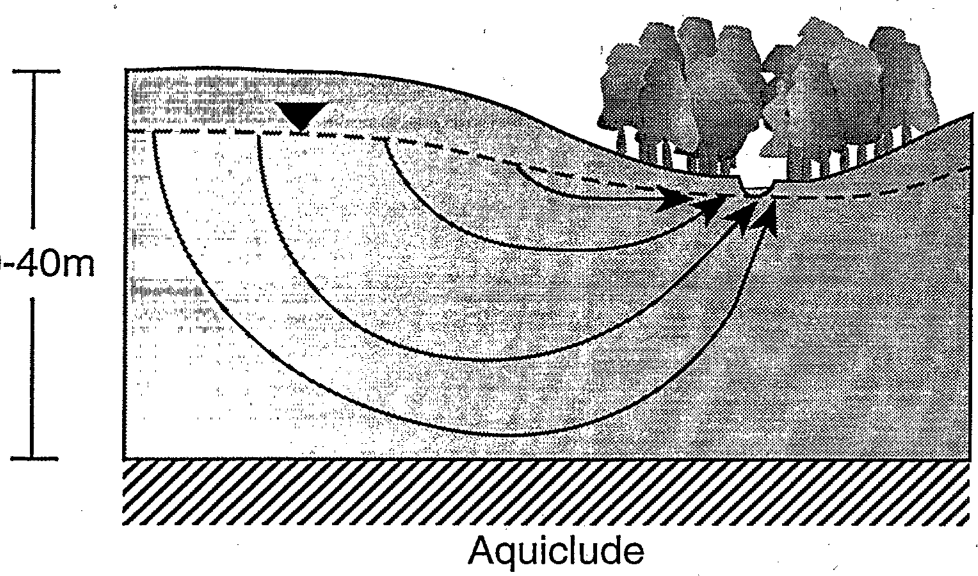

FIGURE 7. Outer Costal Plain—Well-Drained Upland flow system (from Phillips et al., 1993)

7) For a contributing area to RFBS area ratio of about 2:1, the range of sediment and sediment-borne N and P reductions that could be expected under worst-case conditions is about 96% for sediment, 75% for total N and 77% for total P. Most other cases— with a 2:1 area ratio and better upland conservation practices —would be expected to have lower concentrations leaving the RFBS. These numbers are based on the assumption of non-channelized flow through the RFBS.

Because of the lack of quantitative information on RFBS functions in other hydrologic/physiographic/transpiration settings, the more detailed information from ICP settings will be used to guide quantitative estimates for the other settings. The consensus of the scientific panel was that the ICP data represented an upper limit on the functions of essentially unmanaged RFBS. Numerous management options and management factors discussed below could lead to increases in the effectiveness and sustainability of nonpoint pollution control functions. But in general practice, without depending on the management improvements, the effects of RFBS in the ICP would be representative of other systems in the CBW where essentially 100% of excess precipitation moves through an unmanaged RFBS. Where less than 100% of excess precipitation moves through the RFBS, the NPS pollution control effects would be proportionally less.

b. Outer Coastal Plain

1) Well Drained Upland: Because much of the groundwater flow reaches the stream channel through the hyporheic zone, interactions with biologically active soil layers appear to be limited in the Well-Drained Upland (Figure 7). The consensus of the BPJ group was that for Coastal Plain systems, the WDU represented the other end of the spectrum from the ICP. Processing of groundwater-borne NPS pollutants, including nitrate, would be least in the WDU. Based on the present knowledge of these systems, RFBS in the WDU would remove some nitrate from groundwater. This removal function might be enhanced by vegetation management, especially in the Zone 1 area where tree roots could access groundwater discharge. Consensus decisions for the WDU are:

1) Where hydrologic connections between groundwater and biologically active soil layers are made, RFBS in the WDU should have about the same capacity for nitrate removal as in the ICP areas.
2) The Zone 1 vegetation (adjacent to the stream channel) is very important because of potential access to water and pollutants in the hyporheic zone. Zone 1 vegetation should be managed for N uptake and for formation of high organic matter surface soils. Provision of leaf litter and other organic matter to the stream channels may increase denitrification in the channel and hyporheic zone.
3) RFBS in the WDU portion of the Coastal Plain would have about the same capacity to filter sediment and sediment-borne pollutants from surface runoff as RFBS in the ICP.
4) RFBS in the WDU may have higher capacity for removing dissolved chemicals from surface runoff because of higher available storage for infiltrated surface runoff. This function is directly related to lower water tables in the RFBS.
5) Reestablishment of RFBS in the WDU should focus on headwater streams, many of which have been ditched. Enhancement of existing forests along both small and large streams should focus on control of surface runoff and surface-borne pollutants and on management of Zone 1 to intercept nitrate enriched groundwater.

2) Poorly Drained Upland/Surficial Confined: Functions of RFBS in the Outer Coastal Plain hydrologic systems designated Surficial Confined and Poorly Drained Upland are thought to be intermediate between the WDU and the ICP. These flow systems are represented in Figure 8. Specifically, the consensus BPJ on these regions included the following:

1) Potential for nitrate removal is intermediate between WDU and ICP. Generally lower regional groundwater concentrations of nitrate will lead to lower actual removal rates and to less important role for nitrate removal.
2) Agriculture in these regions is commonly associated with artificial drainage which will need to be integrated with RFBS system.
3) Potential for control of sediment and sediment-borne chemicals should be similar to RFBS in the ICP, but actual removal is probably less because of lower loads of surface-borne pollutants.
4) Potential for control of dissolved chemicals in surface runoff may be less than in WDU because of higher water tables and generally less available storage.

c. Tidally Influenced

Tidally influenced areas of the Coastal Plain present unique situations for a number of reasons. First, water and pollutants moving through the terrestrial/aquatic interface move directly into the bay or tidal reaches of streams, providing a direct input of pollutants. Secondly, movement of groundwater through these tidal systems are affected by tidal movements of bay water which serve to restrict discharge from freshwater aquifers. Thirdly, two main types of terrestrial/aquatic interfaces appear to exist, especially for groundwater fluxes. One case is a tidal stream, embayment, or main stem location where a marsh system forms a buffer at the terrestrial/aquatic interface. In areas with marsh, the nitrate removal function of the RFBS is less significant due to groundwater discharge through the marsh being stripped of nitrate in anaerobic marsh sediments. The second case is when the interface does not include the marsh system and discharge takes place from a sand aquifer directly into the bay or tidal stream. The second case is the one that is shown diagrammatically in Figure 9.

The nonpoint pollution control functions of RFBS in tidally influenced areas are dependent on two factors: depth to water table and bank stability. The interaction of water table depth and nitrate removal via denitrification has been discussed extensively in previous sections. Bank stability is a major factor in tidally influenced areas because of wave action, boat wakes, storms, and rising sea level undermining trees at the waters edge. It is likely that in tidal areas with eroding shorelines, trees in a Z one 1 position will contribute to erosion and de-stabilization.

The consensus BPJ on tidal areas of the Coastal Plain include the following:

1) In areas without a tidal marsh at the terrestrial/estuarine interface, nitrate removal should be significant if the water table is within or near the root zone of trees in Zone 2. This removal would be both through direct vegetation uptake and through coupling of vegetation uptake/denitrification in surface soil. Where the water table is consistently below the root zone significant nitrate reduction is unlikely to occur.

2) In areas where shoreline erosion is a problem or potential is high, Zone 1 trees at the water's edge are likely to contribute to shoreline erosion due to undermining of trees and tree fall into tidal waters. If established in these situations, Zone 1 trees need to be put in a position that is not likely to contribute to active erosion, cliff destabilization, or shading of marshes.

3) Functions of Zone 3 for sediment and some nutrient removal should be similar to function in ICP systems.

4) Restoration of RFBS in tidal areas should concentrate on areas with shallow water tables, an absence of tidal wetlands, limited shoreline

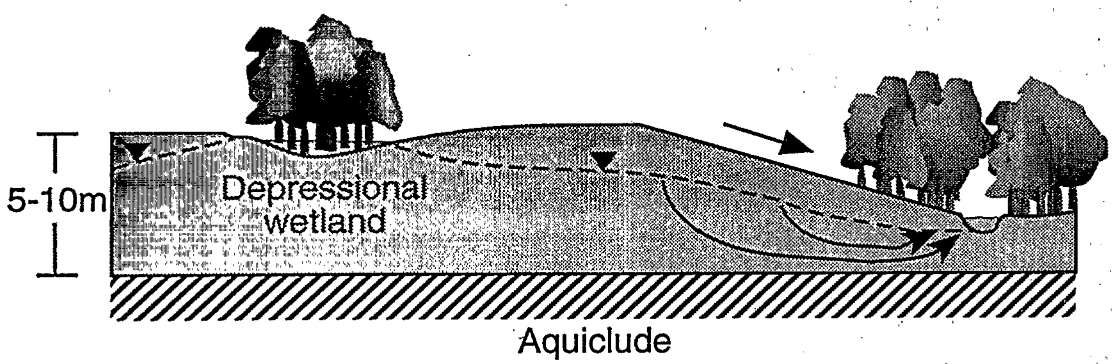

FIGURE 8. *Outer Coastal Plain—Poorly Drained Upland/Surficial Confined flow system.*

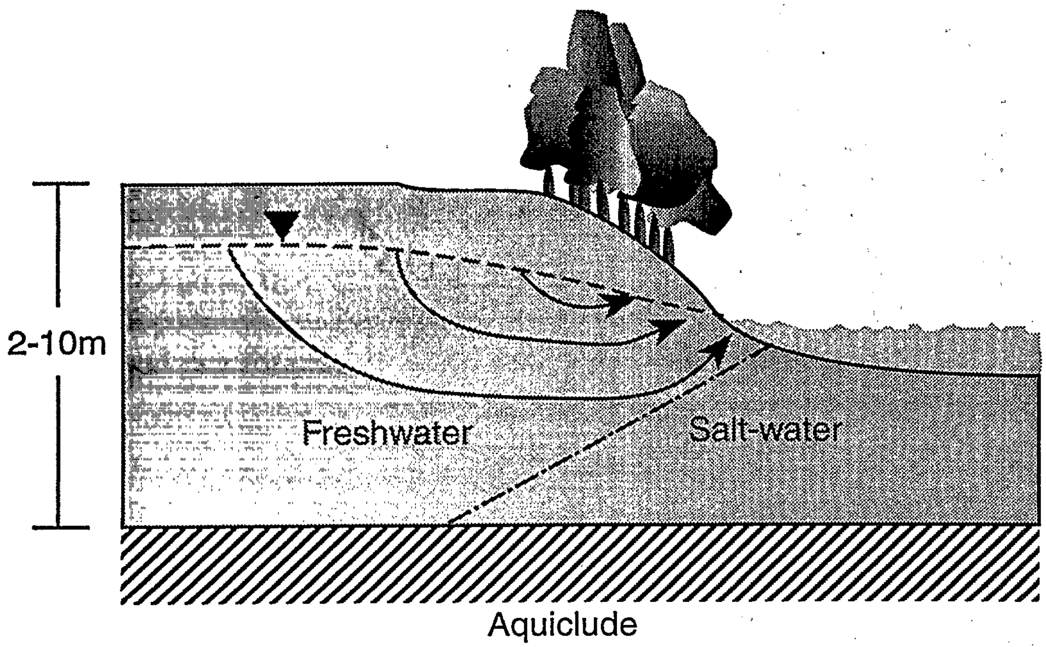

FIGURE 9. *Coastal Plain-Tidal Influenced flow system* (based on Staver and Brinsfield, 1994).

erosion problems and in areas with substantial surface runoff into tidal waters from adjacent land uses.

2. Piedmont

Although substantial work on RFBS has been done in the North Carolina Piedmont and is underway in Piedmont areas of the CBW (See Section II), less information is available in the Piedmont than for the Coastal Plain. The consensus BPJ of the scientific panel was that RFBS in the Piedmont represented a range of conditions for NPS pollution control, depending on both the localized and watershed hydrology and the proportion of excess precipitation which moves through the RFBS. When hydrologic conditions lead to surface runoff to streams and movement of groundwater in or near the root zone of the RFBS, the degree of NPS pollution control should be similar to conditions measured in the North Carolina Piedmont and potentially as effective as the ICP condition. When excess precipitation moves into deeper groundwater and into larger streams through the hyporheic zone, control of groundwater pollutants such as nitrate may be minimal. As described above, baseflow/stormflow separations for Piedmont watersheds should provide a conservative estimate of the quantity of water moving through RFBS.

The first hydrologic condition represented in the Piedmont is in areas with thin soils, direct flow paths to streams, and a large amount of water movement through surface runoff and seepage faces (Figure 10). These conditions are most likely in the Virginia Piedmont in the southern portions of the CBW. Under these conditions, the consensus BPJ are:

1) Nitrate removal would be approximately as effective as in ICP systems. Nitrate removal may be more dependent on vegetation processes because of potential for deeper rooting depth in more aerated soils and the potential for longer residence time for water in Piedmont RFBS.

2) Control of sediment and sediment-borne pollutants in surface runoff should be as effective as ICP and North Carolina Piedmont systems. Control of sediment in surface runoff is likely to be limited by development of concentrated flow channels, especially in steeper RFBS areas of the Piedmont. These areas may require an expanded Zone 2.

3) Control of all sources of P should be represented well by ICP conditions and conditions from North Carolina studies. As in these systems, control should be more effective for sediment-borne P than for dissolved P in either surface runoff or groundwater.

Piedmont areas with deeper soils and saprolite are likely to have longer flow paths and more water entering the stream channel directly from these longer flow paths and the hyporheic zone. These types of Piedmont systems are represented by areas with primarily gneiss/schist bedrock and primarily marble bedrock (Figures 11 and 12). Areas with primarily schist bedrock should have substantial seepage which should be subject to treatment in RFBS.

For Piedmont areas represented in Figures 11 and 12, the Consensus BPJ include:

1) Nitrate removal would be medium in the Piedmont areas with Schist/Gneiss bedrock and should be used to control movement of water in both shallow water table conditions and in seepage areas near streams. Nitrate removal should be least important in Piedmont areas underlain by marble because of movement of groundwater and associated nitrate into regional aquifer systems which will recharge larger streams. This component of groundwater flow is likely to by-pass riparian systems. In both systems, nitrate removal will likely be enhanced by deeply rooted vegetation.

2) Control of sediment and sediment-borne chemicals will depend on management of Zone 3 to reduce the effects of concentrated flow and to protect reestablished forests. Steeper slopes in riparian areas may limit both the sediment filtering capacity and the retention time of water. These conditions may require expanded Zone 3 and/or Zone 2.

3) Control of all sources of phosphorus will be limited by ability to remove dissolved P in surface runoff. Areas with high sediment borne surface runoff P loads should be restored on a priority basis because of potential for controlling these P types.

3. Valley and Ridge/Appalachian Plateau

The Valley and Ridge is represented by larger order

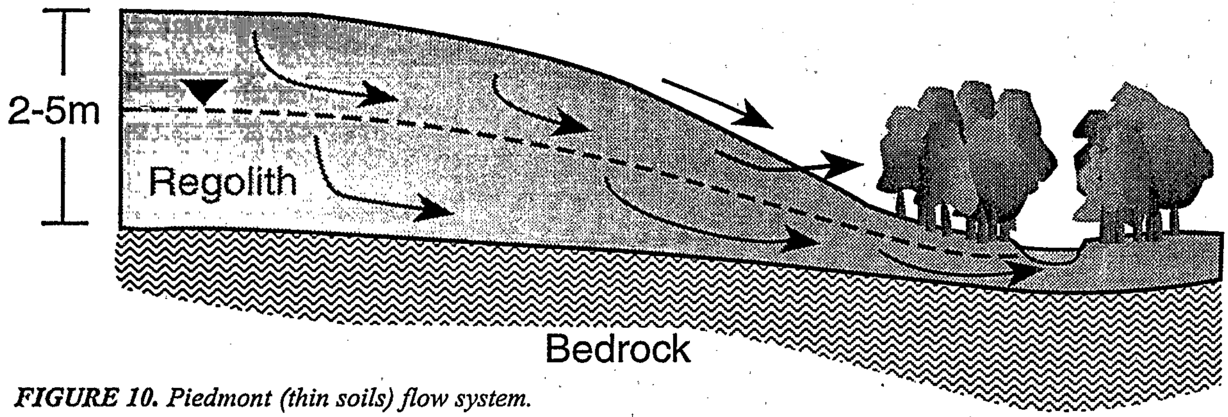

FIGURE 10. Piedmont (thin soils) flow system.

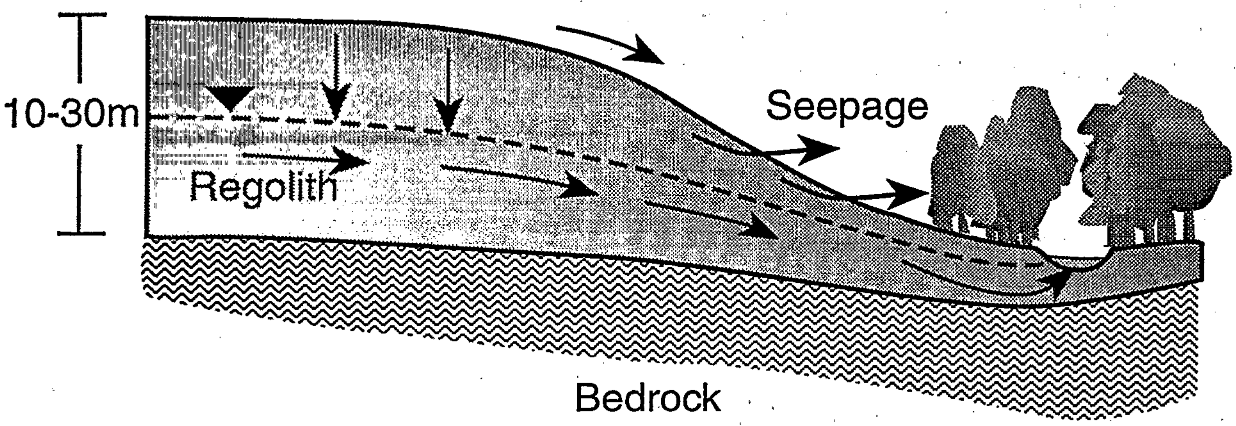

FIGURE 11. Piedmont (schist/gneiss bedrock) flow system.

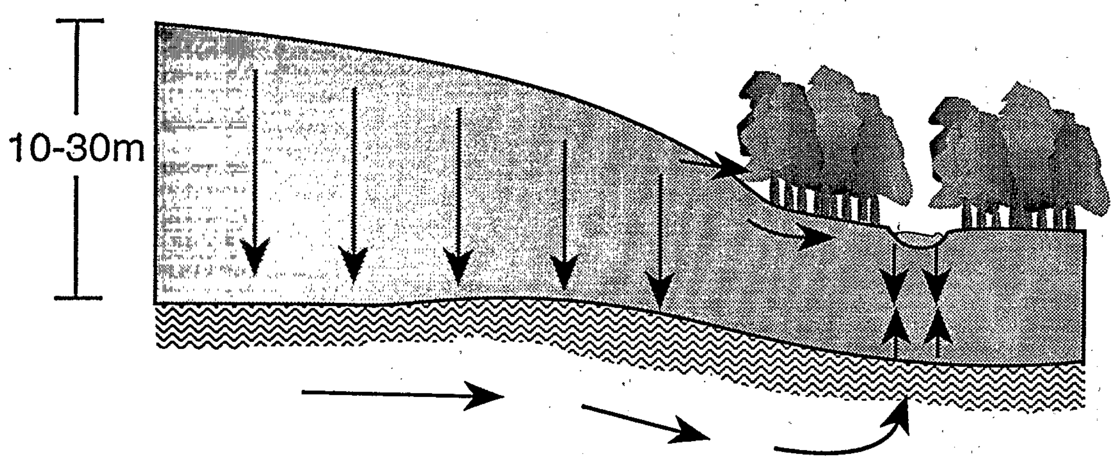

FIGURE 12. Piedmont (marble bedrock)/Valley and Ridge (limestone bedrock) flow system.

streams draining the main valleys with either limestone bedrock (Figure 12) or shale/sandstone bedrock (Figure 13) and by smaller order streams draining the ridges (Figure 14). The situation for sediment and P sources is thought to be similar to the Piedmont hydrologic settings. Nitrate removal will probably show the most variability among the different valley types and with different valley configurations and floodplain extent. Consensus BPJ for larger order streams in the Valley and Ridge for nitrate removal functions are:

1) Valley and Ridge areas with limestone bedrock (Figure 12) will have the least potential for nitrate removal due to most flow going into regional aquifers which are intercepted primarily by major rivers. Seepage areas, springs, and floodplains will have the most potential for nitrate removal. Deep rooted vegetation should be used to control nitrate in these areas.

2) Valley and Ridge with sandstone/shale bedrock (Figure 13) will have more potential for nitrate removal due to less movement of groundwater and nitrate into regional aquifers and the importance and prevalence of seepage areas moving nitrate into biologically active soil horizons. The processing of nitrate is controlled by the presence and size of the floodplain and by the presence of seepage areas and springs. As in other Piedmont and Valley and Ridge settings, deep-rooted vegetation should be used to maximize the potential for N uptake.

3) Nitrate removal from low order streams in both Valley and Ridge and Appalachian Plateau (Figure 14) settings will depend on residence time of water and the presence of

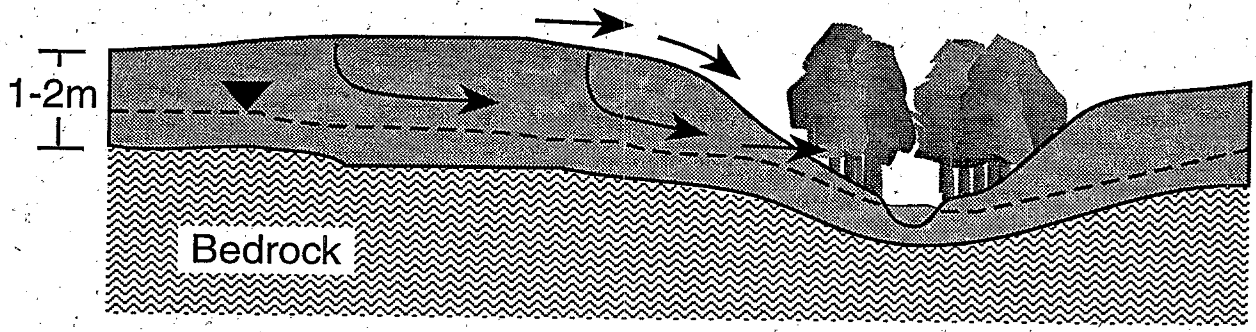

FIGURE 13. Valley and Ridge (sandstone/shale bedrock) flow system.

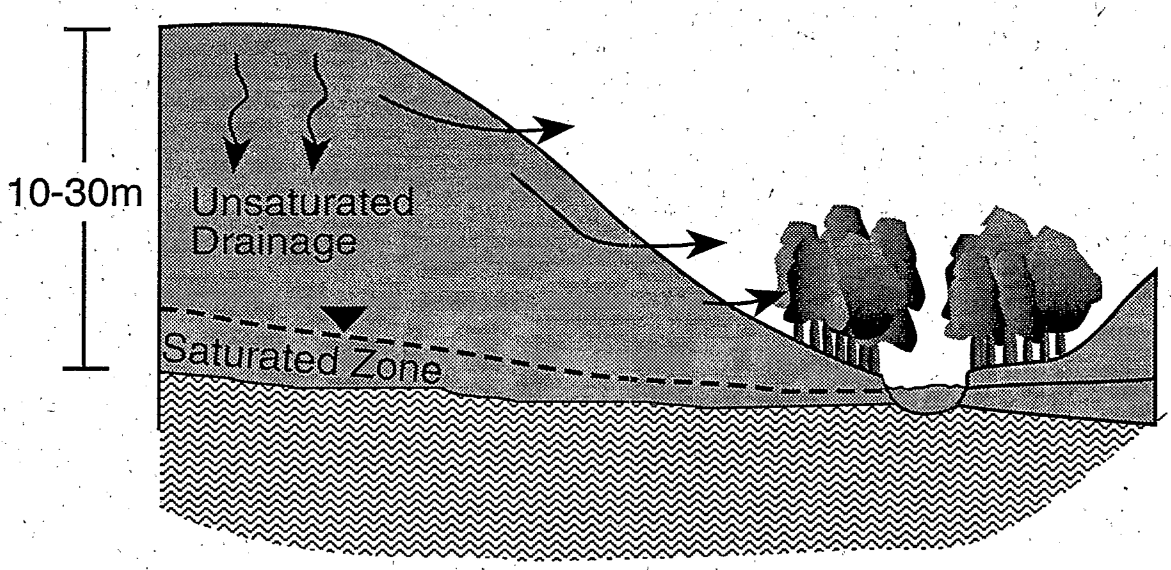

FIGURE 14. Valley and Ridge/Appalachian (low order streams) flow system (based on Mulholland et al., 1990).

seeps and floodplains. In these cases, as in other situations without extensive wetlands, the use of deeply rooted vegetation should enhance nitrate uptake. Because of the limited extent of riparian systems in areas of high relief, Zone 1 will important for nitrate removal in these smaller streams.

C. LOADING RATES AND NONPOINT SOURCE POLLUTION CONTROL

As a nonpoint pollution control practice, Riparian Forest Buffer Systems represent a long-term investment which can change the structure of the agricultural landscape. As a long-term management option, it is quite likely that RFBS will be exposed to a wide range long-term management option, it is quite likely that RFBS will be exposed to a wide range of pollutant loadings due to both interannual variation, and changes in management practices in source areas. Information on how mature RFBS respond to changing pollutant loads is essential to understanding long term sustainability of RFBS.

As discussed above and in Section II, research on some ICP systems indicates that higher rates of nitrate removal would be possible under higher loadings of nitrate. Published studies indicate that this is most likely to be true in areas where denitrification is the primary means of nitrate removal. Given the range in nutrient uptake observed both among different plant species and within the same plant species, it is likely that vegetation uptake will increase with increasing loads, if there is significant hydrologic interaction with vegetation.

Increasing loads of P are likely to be less effectively controlled than increasing loads of N, because of the lack of biological processes to remove or sequester P in the RFBS. If increasing P loads are to be controlled, it will require both effective management of Zones 3 and 2 for sediment removal and management of Zone 2 for infiltration. If dissolved or particulate P can be retained in the root zone, it will be available for both biological and chemical removal processes. If RFBS have some absolute removal potential for P, reducing input loads should increase the efficiency of removal.

Management to control increasing loads of sediment and sediment-borne chemicals will require specific management of Zones 3 and 2 for sediment retention. As described above in Sections I and II, most of the mass of sediment will be deposited in Zone 3 and most of the sediment-borne nutrients will be deposited in Zone 2. Increased sediment loadings to Zone 3 will require increased management to eliminate concentrated flows, remove accumulated sediment especially in berms, and restore the herbaceous vegetation. Increased sediment and sediment-borne chemicals to Zone 2 should lead to higher amounts of chemical deposition in surface litter. As with other dissolved P in surface runoff, the ability of Zone 2 to retain P may be limited, especially under high loadings of dissolved P.

Loading rate/buffer width relationships are only poorly defined, especially for dissolved pollutants. In published studies with water clearly in contact with surface litter or the biologically active root zone, buffers of about 100 feet have been effective for at least sediment and nitrate removal. One of the difficulties in describing these relationships is that increasing pollutant loads may also be accompanied by increasing water volumes in either surface runoff, groundwater, or both. In the presence of increased water movement, denitrification for nitrate removal should be enhanced and sedimentation and infiltration may be decreased. Increased surface runoff and loading of sediment and sediment-borne chemicals can be accommodated by management of Zones 3 and 2 to increase roughness and control channelized flow. Although mass balance approaches, used in Section II may be extrapolated to higher loading rates, they provide only an estimate and may not predict real-world responses.

D. STREAM ORDER/SIZE

Regardless of the size of stream or the hydrologic setting, water moving across the surface or through the root zone of a RFBS should show reduction in either nitrate (groundwater) or sediment and sediment-borne chemical loads reaching the stream. As streams increase in size, the integrated effects of adjacent riparian ecosystems should decrease relative to the overall water quality of the stream. On lower order streams there is greatest potential for interactions between water and riparian areas. For NPS pollution control, the change in impact of RFBS as stream order increases can be estimated based on hydrologic contributions from upstream and from the riparian ecosystem. For first-order streams, the potential impact of the RFBS on chemical load or flow-weighted concentration is directly proportional to the proportion of the excess precipitation from the contributing area which moves through or near the root zone or surface of the RFBS. For all streams above first order, the contributing area is only one source of pollutants, with upstream reaches providing the other source. For second-order and above, the NPS pollution control function of a given RFBS is based on both the proportion of water from the contributing area which moves through the riparian system and the relative sizes of the two potential pollutant loads - upstream sources or adjacent land uses. Clearly, the larger the stream, the less impact a RFBS along a particular stream reach can have on reduction in overall load within that reach. If there are no RFBS upstream from a particular stream reach, the water entering the stream reach is likely to be already contaminated.

On a watershed basis, the higher the proportion of total streamflow originating from relatively short flow-paths to small streams, the larger the potential impact of RFBS. In comparing the potential effectiveness of RFBS among watersheds, drainage density (length of channel per unit area of watershed) should provide a useful starting point. Higher drainage density implies greater potential importance for RFBS in NPS pollution control.

Control of the stream environment is most effective when native vegetation forms a complete canopy over the stream. This is obviously only possible on relatively small streams. The effect of the RFBS on the stream environment is not simply proportional to the amount of the channel which is shaded. As noted above in Section I, besides direct shading of the stream channel, cooling of groundwater recharging streams and provision of bank habitat will occur even on larger streams. Bank habitat, provision of coarse woody debris and provision of leaf detritus remain important functions, regardless of stream size.

E. ESTABLISHMENT AND SUSTAINABILITY

Some aspects of establishment are discussed above. RFBS should be used as part of an integrated land management or conservation system which consists of 1) watershed scale management, 2) NPS pollution management, and 3) active management of the RFBS. In this way, RFBS become part of conservation, stormwater, nutrient and farm management, timber harvest, and other land management planning efforts.

Watershed management is essential to reduce overall pollutant loadings and integrate the riparian area as part of a landscape influenced by upstream hydrology. In a landscape context, RFBS which mimic the natural ecosystems of the area will increase the likelihood of long-term sustainability. Consideration of existing riparian forests and linkage of RFBS as continuous stream corridors is desirable. Source management and land conservation measures are important in conserving natural resources, reducing overall pollution, and limiting stress on the RFBS. These measures, along with maintenance of buffer plantings, are especially important during the establishment phase and in preventing excessive runoff or sediment and nutrient loading beyond the capacity of the buffer. RFBS management such as periodic harvesting, runoff control maintenance, control of invasive plants, etc., is desirable to maximize performance and ensure long term effectiveness. Continued runoff control and protection of Zone 1 functions are essential to maintaining optimum performance in RFBS.

Integration of RFBS within land management helps to prevent some of the primary reasons for "acute" failure such as runoff inputs which exceed the design of the RFBS and cut gullies or channels, or failure to address "chronic" problems such as a gradual decrease in phosphorus retention. Where gullies have formed into or through riparian forests, measures other than flow-spreading in Zone 3, will be necessary to control channelized flow. Because of the commitment of land required for RFBS establishment, the approaches used for establishment and subsequent management should contribute to a RFBS which is sustainable for decades.

At least one sustainability question has been raised relative to each zone of the RFBS. The major sustainability question for Zone 3, discussed in Section I, above is the need to remove accumulated sediment and reestablish herbaceous vegetation periodically. Functions of Zone 3 should be sustainable given proper management of the sediment and vegetation. The other two sustainability questions are closely related to Zones 2 and 1. In most cases, the sustainability of Zone 1 functions will depend on having a Zone 2 which is harvested infrequently. Biomass plantations which require frequent coppicing of trees or grassed Zone 2 areas are likely to expose Zone 1 vegetation to catastrophic failure due to blow down of trees. Zone 2 functions, if dependent on particular types of vegetation, such as deep-rooted species or vegetation specific for high levels of nutrient uptake, will require some management to control invasive species.

IV Research Needs

Research needs are grouped into four general objectives: 1) assessment of existing riparian forest ecosystems relative to the minimum RFBS standards; 2) assessment of the potential for RFBS restoration areas to control NPS pollution; 3) assessment of effectiveness of NPS pollution removal in pilot restoration and enhancement areas; and 4) determination of the effects of management factors on the NPS pollution control functions of restored and enhanced RFBS. Ideally, objectives 1 and 2 would be completed as guidance for pilot restoration and enhancement studies or large scale research projects which would be used as the basis to achieve objectives 3 and 4. If ongoing assessment work related to these first two objectives is done in a timely manner, it will provide substantial guidance to achieve objectives 3 and 4.

The assessment and evaluation of existing riparian forest ecosystems will require the use of remotely sensed data for delineation and classification of riparian forests. Significant progress has been made in assessment of the forest resources of Maryland in a "Comprehensive Forest Resources Inventory for the State of Maryland" undertaken by the Maryland Dept. of Natural Resources (Lade, 1994). The objective of this project was to use Thematic Mapper data to create maps, statistical summaries and digital data sets to describe the location and extent of forest (especially streamside forests) in the state of Maryland. The study was designed for delineation of all forest resources with a minimum mapping unit of 1 acre and a minimum mapping unit of 100 feet for linear forest areas associated with streams. The data are then used in a Geographic Information System to characterize the extent and types or absence of forest in 30 feet and 100 m riparian buffers. This was done to explicitly assist in the identification of potential riparian forest buffer restoration sites for the entire state. The characterization of linear forest should be done at a finer resolution (10 to 20 m) in order to delineate riparian forest buffers of the width recommended in the RFBS specification. Data for these narrower linear forests are needed for the entire CBW in order to characterize the riparian forest resources and forest cover in riparian areas.

One use of the forest inventory will be to overlay other digital layers for further analysis of the relationship of riparian forest buffers to other landscape characteristics (Lade, 1994). The classification scheme developed here could be used as the starting point for an assessment of the potential for existing, enhanced, or restored RFBS to intercept surface runoff or subsurface borne NPS pollutants. Refinement of the classification scheme based on existing and new geohydrology data could be used to produce basin wide maps of the relative potential for control of surface and subsurface borne pollutants. These maps, overlain with maps of riparian buffer vegetation and other data layers such as wetland soils would make it possible to quantify the riparian areas with different potential for NPS pollution control which were available for restoration on a subwatershed basis. Research summarized in this report, as well as forthcoming research results, could be used in conjunction with the mapped information to make quantitative or comparative estimates of the amount of NPS pollution reduction relative to the load reduction goals set for the Chesapeake Bay.

Concurrent with development of a mapping approach is the need to make field assessments of the potential for hydrologic interaction between nonpoint pollutant sources and the RFBS. Reliable indicators of the degree of interaction between groundwater/surface water and the RFBS will be necessary when making field/farm/subdivision/ or watershed scale assessments of nonpoint pollution control potential. Streamflow data from USGS and other sources could be used to assess stormflow/baseflow proportions as a screening technique. Watersheds with higher proportions of stormflow could be targeted for more intensive reconnaissance investigations to determine the potential applicability of RFBS.

The outputs of objectives 1 and 2 should be used to guide the establishment of pilot restoration and enhancement projects and large scale research projects. Only limited objectives can be accomplished in RFBS restoration and management research conducted at the small scales normally associated with agricultural plot research. A number of hypothetical examples can be used to show the potential and the limitations of small scale research in RFBS. For instance, small plots are being used to examine the effects of vegetation management on surface runoff spread evenly through a restored RFBS. These same small plots cannot be used for the study of effects of concentrated flow on the filtering capacity of the RFBS. Similarly, the effects of vegetation management on subsurface flow cannot be studied on small plots. The minimum size for plots to look at long term effects of clear-cutting Zone 2 vegetation is constrained by the ability of trees in adjacent reference areas to put roots into the clear-cut areas which will affect the hydrology of both reference and clear-cut areas.

The ideal scale to accomplish RFBS research should be based on the land uses contributing nonpoint pollution and the hydrology of the system. It may be necessary to conduct work at the watershed scale where accurate streamflow gaging data can be used to assess the effects of RFBS on watershed responses over time. At a minimum, the scale of research is probably that of the representative hillslope. Ideally, integrated research programs at a number of spatial scales would be pursued simultaneously. For instance, a number of hillslope studies with different nonpoint pollution sources might be conducted in one watershed. The hillslopes studies could be used to: 1) examine the effects of differing pollutant loads/sources on similar RFBS; 2) examine the effects of differing RFBS management on similar pollutant loads, or a combination of the two approaches; or 3) examine the effects of differing hydrologic conditions on RFBS functions. These research projects should include sub-objectives to understand the processes responsible for removal of NPS pollution. At the same time, restoration or enhancement of RFBS for significant portions of entire subwatersheds could provide for a comparison with other watersheds without the RFBS restoration/enhancement.

The above discussion of objective 3 amounts to an argument for the integration of research and demonstration projects on RFBS in the CBW and elsewhere. The advantages to the research programs are in both the ability to conduct research at the appropriate scales and the ability to relate the research to "real-world" restorations. The advantage to the demonstration or operational restoration and/or enhancement project is the potential for direct quantification of the water quality benefits of RFBS in different land use/hydrologic/buffer management settings.

A long list of sub-objectives is possible for objectives 3 and 4 of a general research program. Among the potential research topics are: 1) effects of vegetation type and management on sustainability of RFBS; 2) effects of vegetation type and management on NPS pollution control by RFBS; 3) effects of chronic stressors such as long-term N loading and N-saturation on NPS pollution control; 4) effects of acute stressors such as large storms or extremes in temperature or growing season rainfall. For any given size and location of RFBS, the actual degree of NPS pollution control may be dependent on management factors. Although the existing research provides little guidance in this area, management factors are likely to help control the effects of both chronic and acute stressors.

A viable approach to these research needs would be to continue funding for the assessment and mapping work under objectives 1 and 2 while developing/enhancing the coordination between institutions and individuals involved with pilot programs and demonstration projects and institutions and individuals interested in pursuing research associated with these projects.

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